

**Guidelines for
Identifying, Assessing and Managing
Contaminated Sediments in Ontario:
An Integrated Approach**

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Guidelines for Identifying, Assessing and Managing Contaminated Sediments in Ontario: An Integrated Approach

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Preface

Historically, two Ministry of the Environment documents addressed the assessment and management of sediment:

- MOE 1993 – *Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario*. This document provided a set of numerical guidelines for protection of sediment dwelling organisms and guidance on using PSQGs to assess contaminated sediment.
- MOE 1996 – *An Integrated Approach to the Evaluation and Management of Contaminated Sediments*. This document built on the information provided in MOE 1993 by providing additional guidance on assessing contaminated sediment, determining sediment management options, and implementing sediment remediation.

Recently, under the 2002 *Canada-Ontario Agreement Respecting the Great Lakes Basin Ecosystem* (COA), the Ministry and Environment Canada developed a harmonized framework for assessing contaminated sediments in the Great Lakes (COA 2007, *Canada-Ontario Decision Making Framework for Assessment of Great Lakes Contaminated Sediment*). This COA sediment assessment framework builds on Ontario's existing ecosystem approach to sediment assessment; it identifies all possible sediment assessment outcomes based on four lines of evidence (sediment chemistry, toxicity, benthos alteration, and biomagnification potential) and provides more specific direction on next steps in making sediment management decisions.

While developed specifically for use in the Great Lakes, the Ministry recognized that the COA sediment framework can be applied to assess contaminated sediments province-wide. This document incorporates the COA sediment assessment framework with existing Ministry guidance on assessing and managing contaminated sediments (MOE 1993 and MOE 1996) into a single integrated Ministry guidance document for identifying, assessing, and managing contaminated sediments in Ontario. To avoid confusion, the previous Ministry sediment guidance documents have been replaced with this integrated document.

Minor modifications were made to portions of the previous Ministry guidance documents to either update sections with current information (e.g., update the section on legislation; replace Open Water Disposal Guidelines with Lakefill Guidelines) or to make the material easier to read (e.g., some sections were rearranged). However, no changes were made to the Provincial Sediment Quality Guidelines (PSQGs) or to the process used to develop them. For this document, the only significant change to previous MOE guidance is the addition of the COA sediment assessment framework. The Ministry is planning to review PSQGs and the protocol used to derive them. This document may be revised in the future based on the results of this review.

Minor modifications were also made to the COA sediment assessment framework to ensure that the information is consistent with existing Ministry regulations and environmental programs. These changes did not change the process outlined in the COA sediment assessment framework.

Public consultation on the methodologies and guidelines referred to in this integrated sediment guidance document have occurred via the Environmental Registry as follows:

- An Integrated Approach to the Evaluation and Management of Contaminated Sediments (MOE 1996) was posted on the Environmental Registry with a public comment and review period starting April 19, 1995 through to May 19, 1995 (see EBR # PA5E0016)
- Canada-Ontario Agreement Contaminated Sediment Assessment Decision-Making Framework (COA 2007) was posted on the Environmental Registry with a public comment and review period starting November 21, 2006 through to January 20, 2007 (see EBR # PA06E0007)
- Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario (MOE 1993) was not posted on the Environmental Registry as it predates the establishment of the registry. However, it was reviewed by the Advisory Committee on Environmental Standards, an independent advisory body in place at that time to advise the Ministry of the Environment on environmental standards.

Executive Summary

Contaminated sediment is a major environmental concern in many areas of Ontario, especially the Great Lakes (IJC 1985). Persistent toxic substances that have accumulated in bottom sediments from industrial, municipal and non-point sources are a threat to the survival of bottom-dwelling (benthic) organisms and their consumers, and can impair the quality of the surrounding water. In order to deal effectively with sediment contamination problems, environmental managers need to determine levels of contaminants that do not pose a risk to sediment-dwelling organisms and their consumers (or to other water uses), from levels of contaminants that are potentially detrimental.

It has long been recognized that sediment chemistry alone is not sufficient to accurately predict adverse biological effects. Thus, sediment chemistry is usually only used as an initial screening tool for determining if higher 'effects based' tiers are needed. The use of biological endpoints, such as information on sediment toxicity, the benthic community and/or biomagnification potential are often needed to properly assess the potential ecological and human health risk posed by contaminated sediment. This document provides guidance on identifying, assessing, and managing contaminated sediments in Ontario using both chemical and biological information. It integrates the guidance provided in two earlier Ministry documents for assessing and managing contaminated sediment (MOE 1993, MOE 1996) with a decision-making framework for assessing contaminated sediments developed jointly by the Ministry and Environment Canada (COA 2007).

This document is comprised of 3 parts:

1. Identifying potential sediment contamination (from MOE 1993, *Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario*).
2. Assessing the potential for contaminated sediments to impair the aquatic environment (from COA 2007, Canada-Ontario Decision Making Framework for Assessment of Great Lakes Contaminated Sediment).
3. Managing contaminated sediments to reduce contamination and adverse impacts to the aquatic environment (from MOE 1996, *An Integrated Approach to the Evaluation and Management of Contaminated Sediments*).

Potential sediment contamination is identified by comparing concentrations of substances in sediment to the Provincial Sediment Quality Guidelines (PSQGs); a set of numerical guidelines for the protection of sediment dwelling organisms. When PSQGs are exceeded, additional biological assessments, such as sediment toxicity tests and other biological effects (e.g. benthic community and biomagnification potential), are evaluated. Assessment of contaminated sediment relies on the approach provided in the COA Sediment Framework (COA 2007). The framework considers four lines of evidence simultaneously (sediment chemistry, toxicity, benthos alteration, and

biomagnification potential), and this information is then assessed in a decision-making matrix whereby the need for management actions can be identified. The assessment component of the 1996 MOE guidance document and the COA framework both consider the same lines of evidence. However, the COA framework is more structured and transparent. The sequential process to assessing sediment contamination outlined in the COA framework has been adopted and incorporated into this document. The decision matrix, which is compiled following the assessment of the four lines of evidence, allows areas to be identified that require no additional action, further assessment, or specific management decisions.

Prior to any remedial action, permits and approvals may be required under different legislation. An overview of the legislation which may impact management decisions, as well as a brief overview of some remedial options, are provided based primarily on information contained in MOE 1996 but updated to reflect new and amended legislation.

Acknowledgements

Many ministry staff contributed their knowledge, expertise and time to review this document and make useful comments and suggestions. The two MOE documents that make up an integral part of this document were originally developed by R. Jaagumagi, D. Persaud and A. Hayton. The sediment assessment portion of this document, derived from the COA sediment assessment framework, was a result of input from staff from provincial and federal government agencies, as well as external agencies (including academia, the private sector, and other governments). The final COA document was prepared by P.M. Chapman, under the guidance of J. Anderson (Environment Canada) and D. Boyd (MOE), and the members of the COA Sediment Task Force.

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1 Introduction

1.1 Background

Contaminated sediments are a major environmental concern in many areas of Ontario, especially the Great Lakes (IJC 1985). Persistent toxic substances that have accumulated in bottom sediments from industrial, municipal and non-point sources are a threat to the survival of bottom-dwelling (benthic) organisms and their consumers, and can also impair the quality of the surrounding water.

Sediments contaminated by such substances have become a critical problem for environmental managers. In order to deal effectively with sediment contamination problems, environmental managers need to determine levels of contaminants that do not pose a risk to sediment-dwelling organisms and their consumers (or to other water uses), from levels of contaminants that are potentially detrimental.

The need for biological effects-based guidelines for the evaluation of sediment is well recognized. Current sediment related issues are much broader than those identified in the early 1970's and knowledge based on information accumulated over the last decade or so (e.g., 1980s) requires that strategies be developed to manage contaminated sediment. Guidelines for the evaluation of sediment must provide the basis for determining when sediments are considered clean, what levels of contamination are acceptable in the short-term, and when contamination is severe enough to warrant significant remedial action.

It has long been recognized that sediment chemistry alone cannot be used to predict biological effects. Thus, sediment chemistry is often only used as an initial screening tool for determining if higher 'effects based' tiers are needed. The use of biological endpoints, such as sediment bioassays, benthic community evaluation and biomagnification potential are crucial in the assessment of sediments for dredging, cleanup or monitoring.

1.2 Purpose

In the 2002 Canada-Ontario Agreement (COA) Area of Concern (AOC) Annex, Result 4 "*Management Strategies for Contaminated Sediment*", both Canada and Ontario committed to developing a 'risk-based decision-making framework' for the assessment of contaminated sediments. COA Annex AOC: Result 4 identifies the need to develop management strategies for contaminated sediment in the Great Lakes. Towards that goal, the Ministry of the Environment (MOE) and Environment Canada have developed a sediment assessment tool.

Two Ministry guidance documents were historically available with respect to the assessment and management of contaminated sediment:

- MOE 1993 – *Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario* (PSQGs). This document provided a set of numerical guidelines for protection of aquatic biota: the No-Effect Level (NEL), the Lowest Effect Level (LEL) and Severe Effect Level (SEL). The NEL are based on the related Provincial Water Quality Objectives (PWQO) and the LEL and SEL are based on co-occurrence of chemical and biological information. Some guidance on using PSQGs to assess contaminated sediment was also provided in this document.
- MOE 1996 – *An Integrated Approach to the Evaluation and Management of Contaminated Sediments* built on the information provided in MOE 1993 by providing additional guidance on assessing contaminated sediment and information on managing sediment contamination.

The COA Sediment Decision-Making Framework (COA 2007) provides one consistent and harmonized approach to assess contaminated sediment. The framework provides a decision making framework that considers four lines of evidence simultaneously (sediment chemistry, toxicity, benthos alteration, and biomagnification potential), and identifies the need for management actions based on observed exceedances/effects in the different lines of evidence. The previous MOE 1996 document and the COA sediment assessment framework both consider the same lines of evidence. However, the COA framework is more structured and transparent.

This document provides guidance on identifying, assessing, and managing contaminated sediments in Ontario using both chemical and biological information. It integrates the guidance provided in two earlier Ministry documents for assessing and managing contaminated sediment (MOE 1993, MOE 1996) with a decision-making framework for assessing contaminated sediments developed jointly by the Ministry and Environment Canada (COA 2007). This document comprises 3 parts: (1) identifying sediment contamination; (2) assessing its potential to impair the aquatic environment; and, (3) managing the problem.

Section I: IDENTIFICATION

The identification of contaminated sediment is achieved through the comparison of sediment chemistry to numerical sediment quality guidelines as previously provided in MOE 1993, *Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario*. This section provides a description (or listing) of inorganic and organic Provincial Sediment Quality Guidelines (PSQGs) and their application for assessing sediment contamination.

2. Provincial Sediment Quality Guidelines

2.1 Overview

The purpose of the Provincial Sediment Quality Guidelines is to protect the aquatic environment by setting safe levels for metals, nutrients (substances which promote the growth of algae) and organic compounds.

The guidelines establish three levels of effect - No Effect Level, Lowest Effect Level and Severe Effect Level. The Lowest Effect Level and Severe Effect Level are based on the long-term effects which the contaminants may have on the sediment-dwelling organisms. The No Effect Level is based on levels of chemicals which are so low that significant amounts of contaminants are not expected to be passed through the food chain.

The three levels of effect are:

The No Effect Level: The No Effect Level (NEL) indicates a concentration of a chemical in the sediment that does not affect fish or sediment-dwelling organisms. At this level negligible transfer of chemicals through the food chain and no effect on water quality is expected. Sediment meeting the NEL are considered clean.

The Lowest Effect Level: The Lowest Effect Level (LEL) indicates a level of contamination that can be tolerated by the majority of sediment-dwelling organisms. Sediments meeting the LEL are considered clean to marginally polluted.

The Severe Effect Level: The Severe Effect Level (SEL) indicates a level of contamination that is expected to be detrimental to the majority of sediment-dwelling organisms. Sediments exceeding the SEL are considered heavily contaminated.

The protocol and rationale for setting PSQGs is provided in Appendix A. PSQGs are provided below in the following tables for metals and nutrients (Table 1), PCBs and organochlorine pesticides (Table 2a), and Polycyclic Aromatic Hydrocarbons (Table 2b). Background concentrations for metals and organic compounds are provided on Table 3 and 4, respectively.

Table 1: Provincial Sediment Quality Guidelines for Metals and Nutrients^a

	No Effect Level	Lowest Effect Level	Severe Effect Level
METALS			
Arsenic	- ^b	6	33
Cadmium	-	0.6	10
Chromium	-	26	110
Copper	-	16	110
Iron (%)	-	2	4
Lead	-	31	250
Manganese	-	460	1100
Mercury	-	0.2	2
Nickel	-	16	75
Zinc	-	120	820
NUTRIENTS			
TOC(%) ^c	-	1	10
TKN ^c	-	550	4800
TP ^c	-	600	2000

^a. Values in µg/g dry weight unless otherwise noted (µg/g = ppm). Values less than 10 have been rounded to one significant digit. Values greater than 10 have been rounded to two significant digits except for round numbers which remain unchanged (e.g., 400).

^b. “-” denotes insufficient data/no suitable method

^c. TOC – Total Organic Carbon; TKN – Total Kjeldahl Nitrogen; TP – Total Phosphorus

Table 2a: Provincial Sediment Quality Guidelines for PCBs and Organochlorine Pesticides^a

Compound	No Effect Level	Lowest Effect Level ^b	Severe Effect Level ^b (µg/g organic carbon)*
Aldrin	- ^c	0.002	8
BHC	-	0.003	12
α-BHC	-	0.006	10
β-BHC	-	0.005	21
γ-BHC	0.0002	(0.003) ^{d,e}	(1) ^f
Chlordane	0.005	0.007	6
DDT (total)	-	0.007	12
op + pp-DDT	-	0.008	71
pp-DDD	-	0.008	6
pp-DDE	-	0.005	19
Dieldrin	0.0006	0.002	91
Endrin	0.0005	0.003	130
HCB	0.01	0.02	24
Heptachlor	0.0003	-	-
Heptachlor epoxide	-	0.005 ^e	5 ^f
Mirex	-	0.007	130
PCB (total)	0.01	0.07	530
PCB 1254 ^g	-	(0.06) ^e	(34) ^f
PCB 1248 ^g	-	(0.03) ^e	(150) ^f
PCB 1016 ^g	-	(0.007) ^e	(53) ^f
PCB 1260 ^g	-	(0.005) ^e	(24) ^f

^a. Values in µg/g dry weight unless otherwise noted (µg/g = ppm). Values less than 10 have been rounded to one significant digit. Values greater than 10 have been rounded to two significant digits except for round numbers which remain unchanged.

^b. Lowest Effect Levels and Severe Effect Levels are based on the 5th and 95th percentiles respectively of the Screening Level Concentration (SLC) except where noted otherwise (See Appendix A).

^c. “-” denotes insufficient data/no suitable method

^d. Values in round brackets “()” are tentative guidelines

^e. 10% SLC

^f. 90% SLC

^g. Analyses for PCB Aroclors are not mandatory unless specifically requested by MOE.

* Numbers in this column are to be converted to bulk sediment values by multiplying by the actual TOC concentration of the sediments (to a maximum of 10%). For example, analysis of a sediment sample gave a PCB value of 30 ppm and a TOC of 5%. The value for PCB in the Severe Effects column is first converted to a bulk sediment value for sediment with 5% TOC by multiplying 530 x 0.05 = 26.5 ppm as the Severe Effect Level guideline for that sediment. The measured value of 30 ppm is then compared with this bulk sediment value and is found to exceed the guideline.

Table 2b: Provincial Sediment Quality Guidelines for Polycyclic Aromatic Hydrocarbons^a

Compound ^b	No Effect Level	Lowest Effect Level ^c	Severe Effect Level ^c (µg/g organic carbon)*
Anthracene	- ^d	0.220	370
Benzo[a]anthracene	-	0.320	1,480
Benzo[k]fluoranthene	-	0.240	1,340
Benzo[a]pyrene	-	0.370	1,440
Benzo[g,h,i]perylene	-	0.170	320
Chrysene	-	0.340	460
Dibenzo[a,h]anthracene	-	0.060	130
Fluoranthene	-	0.750	1,020
Fluorene	-	0.190	160
Indeno[1,2,3-cd]pyrene	-	0.200	320
Phenanthrene	-	0.560	950
Pyrene	-	0.490	850
PAH (total) ^e	-	4	10,000

^a. Values in µg/g dry weight unless otherwise noted (µg/g = ppm).

^b. Guidelines could not be calculated for Acenaphthene, Acenaphthylene, Benzo[b]fluorene and Naphthalene due to insufficient data.

^c. Lowest Effect Levels and Severe Effect Levels are based on the 5th and 95th percentiles respectively of the Screening Level Concentration (SLC) except where noted otherwise (See Appendix A).

^d. “-” denotes insufficient data to calculate guideline.

^e. PAH (total) is the sum of 16 PAH compounds: Acenaphthene, Acenaphthylene, Anthracene, Benzo[k]fluoranthene, Benzo[b]fluorine, Benzo[a]anthracene, Benzo[a]pyrene, Benzo[g,h,i]perylene, Chrysene, Dibenzo[a,h]anthracene, Fluoranthene, Fluorene, Indeno[1,2,3-cd]pyrene, Naphthalene, Phenanthrene, and Pyrene.

* Numbers in this column are to be converted to bulk sediment values by multiplying by the actual TOC concentration of the sediments (to a maximum of 10%). For example, analysis of a sediment sample gave a B[a]P value of 30 ppm and a TOC of 5%. The value for B[a]P in the Severe Effects column is first converted to a bulk sediment value for sediment with 5% TOC by multiplying 1443 x 0.05 = 72 ppm as the Severe Effect Level guideline for that sediment. The measured value of 30 ppm is then compared with this bulk sediment value and is found to not exceed the guideline.

Table 3: Background Sediment Concentrations for Metals^a

Metal	Background (µg/g)
Arsenic	4
Cadmium	1
Chromium	31
Copper	25
Iron (%)	3
Lead	23
Manganese	400
Mercury	0.1
Nickel	31
Zinc	65

^a Values are based on analysis of Great Lakes pre-colonial sediment horizon

Table 4: Background Sediment Concentrations for Organic Compounds^a

Metal	Background (µg/g dry wt.)
Aldrin	0.001
α-BHC	0.001
β-BHC	0.001
γ-BHC	0.001
Chlordane	0.001
DDT (total)	0.010
op + pp-DDT	0.005
pp-DDD	0.002
pp-DDE	0.003
Dieldrin	0.001
Endrin	0.001
HCB	0.001
Heptachlor	0.001
Heptachlor epoxide	0.001
Mirex	0.001
PCB (total)	0.020

^a Values are based on the highest of the Lake Huron or Lake Superior mean surficial sediment concentration

2.2 Application of Sediment Quality Guidelines

The Provincial Sediment Quality Guidelines (PSQGs) shown in Tables 1 and 2 provide the basis for all sediment (or potential lakefill materials to be placed in water) evaluations in Ontario. The guidelines pertain mainly to activities within the aquatic environment and adherence to them is not to be construed as exemption from the requirements of other guidelines, policies, or regulations of this Ministry or other agencies (e.g., the placement of contaminated sediment at an upland site or facility will be subject to the requirements of the Ministry's Waste Management Regulations). The PSQGs can be used in making decisions on a number of sediment-related issues ranging from prevention of sediment contamination to remedial action for contaminated sediment. Issues to be addressed include, but are not limited to, the following:

- As one line of evidence for assessing contaminated sediments as described in the sediment decision making framework (see Section II).
- Determining appropriate action with regard to sediment clean-up in areas with historic sediment contamination, as well as other areas of potential impact to the environment.
- Determining fill quality for lakefilling associated with shoreline development programs.
- Establishing the chemical suitability of substrate material for the restoration of benthic habitat.
- Determining the appropriate degree of sediment clean-up as a result of chemical spills or unauthorized discharge.

2.3 The Evaluation Process

Initial evaluation of bottom sediment or fill material is conducted by comparing the chemical concentrations of the material to the appropriate parameter values listed in Tables 1, 2a and 2b, and where required Tables 3 and 4, based on the conditions described in Section 2.3.1.

2.3.1 General Conditions Governing Evaluation

- (a) Material will be tested by bulk sediment analyses and results reported on a dry weight basis, ideally as per MOE analytical methods (MOE 2004a,b,c; 2005a,b,c), or MOE approved equivalent analytical procedures.
- (b) When comparing analytical results with the PSQGs, the results will be rounded as follows: if the reported value is less than ten, it will be rounded to one significant digit. Values greater than 10 will be rounded to two significant digits.

e.g.	Reported Value	Rounded Value
<10	1.78	2
	0.0364	0.04
	0.0052	0.005
>10	10.827	11
	128.4	130

- (c) If all parameter values for a given material are at (or below) the NEL (if available), that material passes the guideline and it is anticipated that the material will have no adverse chemical effects on aquatic life or water quality.
- (d) If a single parameter value for a given material, based on a sampling program, exceeds the NEL but is below the LEL, the material fails the NEL and would be considered as having a negligible potential to impair the aquatic environment.
- (e) If a single parameter value for a given material, based on a sampling program, is at or above the LEL, that material fails the guideline and it is anticipated that such material may have an adverse effect on some benthic biological resources. If all values are below the LEL, no significant effects on benthic biological resources are anticipated.
- (f) If any single parameter value for a given material, as determined by a sampling program, is at or above the SEL, that material is considered highly contaminated and will likely have a significant effect on benthic biological resources.
- (g) The Ministry recognizes that in an area as geologically diverse as Ontario, local natural sediment levels of metals may vary considerably and in certain areas, such as wetlands, the organic matter content and nutrient levels may be naturally high.

Metals: In areas where local background levels are above the LEL, the local background level will form the practical lower limit for management decisions. In some waterbodies, surficial sediments upstream of all discharges may be acceptable for calculation of background values. Where it cannot be shown that such areas are unaffected by local discharges, the pre-colonial sediment horizon can be used. Site specific background for metals is calculated as the mean of 5 replicate samples from surficial sediment that has not been directly affected by human activity or from the 'pre-colonial' sediment horizon. The calculations are described in Appendix A of this document. Alternatively, the mean background values for the Great Lakes Basin as presented in Table 3 may be used.

Nutrients: Areas of high natural organic matter content, such as marshes and other types of wetlands, can be readily distinguished from those resulting from anthropogenic sources. In such cases, for the nutrients listed in Table 1, the local background would serve as the practical lower limit for management action.

- (h) It is also recognized that long-range sources such as atmospheric deposition have contributed to accumulation of organic compounds in areas remote from any specific source. Therefore, in those areas where specific sources cannot be determined, the practical lower limit for management action is the Upper Great Lakes deep basin surficial sediment concentration. These have been defined for a number of organic compounds and are presented in Table 4.

2.3.2 Specific applications

If the sediment concentration exceeds the Lowest Effect Level, then the concentration is compared with the local background values for that parameter. Background values can be derived from physically contiguous areas that are unaffected by point-source discharges, or if these do not exist, then from the "pre-colonial" sediment horizon. The latter would represent background levels in existence before European colonization of the area and is generally considered as the area below the *Ambrosia* pollen horizon. In those instances where local values are not available, the concentration may be compared to the background values listed in Tables 4. These are based on values from the Great Lakes and may not be applicable to inland sites.

If the sediment concentration is below the natural background then no further management decisions need to be considered. In areas where contaminants in sediment are at or above the Severe Effect Level, the sediment is deemed to be highly contaminated. When there are exceedances of the LEL or SEL, further testing and the development of a management plan may be required.

Section II: ASSESSMENT

3 Sediment Assessment – the sediment decision-making framework

The assessment of contaminated sediments in Ontario is based on the Canada-Ontario Decision Making Framework for Assessment of Great Lakes Contaminated Sediment (COA, 2007). The COA Sediment Decision-Making Framework builds on previous MOE guidance to assess contaminated sediment (MOE 1993, *Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario* and MOE 1996, *An Integrated Approach to the Evaluation and Management of Contaminated Sediments*). The COA framework provides a consistent and harmonized approach to assessing contaminated sediment, is more structured and transparent in how it assesses information, and provides more specific direction on next steps in making sediment management decisions. The decision matrix, which is compiled following the assessment of the four lines of evidence, allows areas to be identified that require no additional action, further assessment or management decisions.

3.1 Guidance for implementation

There are four general guidance “rules” for the use in the implementation of the framework:

1. Sediment chemistry data will not be used alone for remediation decisions except for two cases:
 - (i) Where contamination is such that adverse biological effects are likely and the costs of further investigation outweigh the costs of remediation, and there is agreement to act instead of conducting further investigations (Wenning and Ingersoll, 2002).
 - (ii) Where sites are subject to regulatory action.
2. Any remediation decisions will be based primarily on biology, not chemistry since chemical PSQGs (or other criteria in the absence of a PSQG value) are not clean-up numbers by themselves, and need to be used in a risk assessment framework.
3. Lines of Evidence (LOE, e.g., laboratory toxicity tests, models) that contradict the results of properly conducted field surveys with appropriate power to detect changes (e.g., see Environment Canada, 2002) should be considered incorrect (Suter, 1996), to the extent that other LOE are not indicative of adverse biological effects in the field.
4. When the impacts of a remedial action will result in more environmental harm than leaving the contaminants in place, the remedial action should not be implemented (USEPA, 1998).

3.2 Framework

The framework is a tiered approach, and proceeds through a series of sequential steps; the rationale for each step is provided. However, different steps do not need to be completed separately; two or more steps can (and in some cases should) be completed jointly (i.e., where this will reduce overall time and costs related to sampling and analysis). For example, if available data are insufficient to rule out management action, sediment toxicity tests may be conducted before chemical analyses are conducted for all chemicals with a corresponding guideline or criterion. If toxicity tests show that the sediment is not toxic, and bioaccumulative substances have historically not been observed in the area, there would be no reason to conduct a costly chemical analysis.

The framework is therefore linear in terms of thought processes; however, that linearity does not necessarily have to be followed in actions such as sample collections or analyses. For example, initial field sampling can involve all possible LOEs (e.g.,

sediments for chemical analyses and toxicity testing; benthos for chemical analyses and taxonomy) with the recognition that, while samples for chemical analyses and taxonomy can be archived, those for toxicity testing cannot be archived and should be tested as soon as possible and no later than 8 weeks following collection (EPA/USACOE, 1998).

The framework is conceptually divided into a series of Steps and Decisions that correspond to different ERA tiers. Screening Assessment (Section 4.2.1) comprises Steps 1-3 and Decisions 1-2. Preliminary Quantitative Assessment (Section 4.2.2) comprises Steps 4-5 and Decisions 3-4. Detailed Quantitative Assessment (Section 4.2.3) comprises Step 6 and Decision 5. Step 7 and Decision 6 deal with deeper sediments (i.e. greater than 10cm depth). The framework is illustrated schematically in its entirety in Figure 1 and in terms of the different ERA tiers at the start of Sections 4.2.1 (Figure 2), 4.2.2 (Figure 3), 4.2.3 (Figure 4), and 4.2.3.2 (Figure 5). The individual steps and the decisions made (and rationale) are described in detail in the sections that follow.

“Due to the complexity involved in evaluating contaminated sediment, it is essential that scientists with strong expertise in sediment chemistry (chemical fate, transport and speciation), sediment toxicity testing, benthic community assessment, food chain effects and environmental statistics assist stakeholder groups in the interpretation of the data. This is especially important in determining differences or effects of sediment contamination compared to reference conditions.” (MOE, 1996)

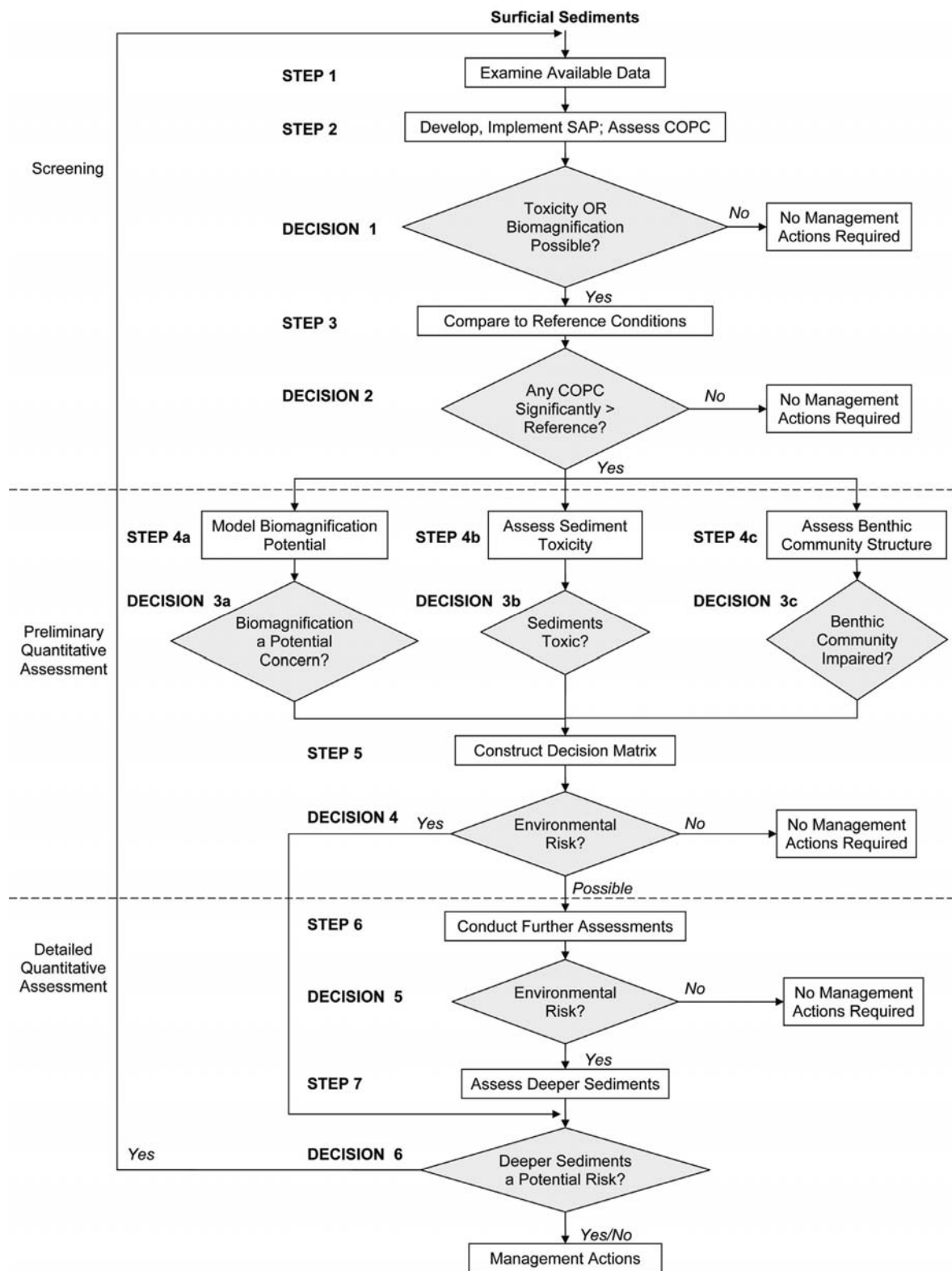


Figure 1. Decision-Making Framework for Contaminated Sediments. For Explanations of Acronyms, Steps and Decisions, see Text.

3.2.1 Screening

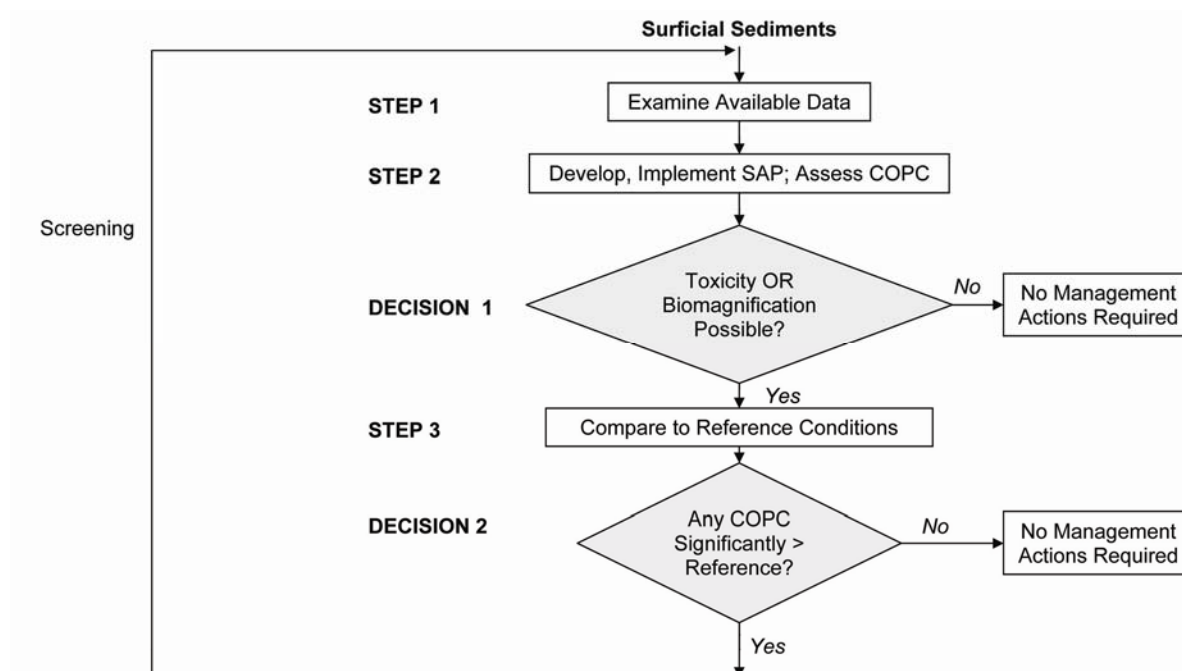


Figure 2. Initial Screening Assessment (Steps 1-3, Decisions 1-2). Conservative (worst case) assumptions are used to screen out locations and substances that are clearly not of concern and to focus on those that may be of concern.

3.2.1.1 *Step 1: Examine available data*

Examine all readily available data for the site (see Section 4.1: Site Definition), reports and information to determine:

- Contaminants of potential concern (COPC: Section 4.2) and their concentrations at surface (e.g., < 10cm) and at depth (e.g., > 10cm);
- Receptors of potential concern (ROPC; i.e. the organisms that may be affected by COPC: Section 4.3); this information may also assist in selection of toxicity test species should the standard MOE protocols (Bedard *et al.* 1992; Environment Canada 1997a,b; ASTM 2000) be deemed inappropriate (see Section 3.2.2.2);
- Exposure pathways (by which COPCs may reach ROPCs);
- Any human health consumption advisories;
- Sediment stability;
- Appropriate assessment endpoints (what is to be protected, e.g., benthos: organisms living in the sediments – see Section 4.4);
- Measures of effect and the level of any effects determined (what is actually measured, e.g., for benthos: species diversity, abundance, dominance – see Section 4.4);

- Appropriate reference areas/locations and their characteristics (see Section 4.5).

Determine whether the site (defined in Section 4.1) has a high level of environmental sensitivity (based on habitat, not land use), and whether contamination is only from off-site sources. A site is defined as the area under investigation which, dependent on size, COPCs and other considerations, will generally require multiple samples to assess any environmental impact. Develop an initial Conceptual Site Model (CSM – showing the interrelationships of COPCs and ROPCs – see Section 4.6), which will be updated as more information becomes available through further investigation.

Information gathered should consider not only surficial sediments (to about 10 cm depth), which are the initial focus, as this is where the majority of sediment-dwelling organisms live, but also deeper sediments and their contamination level and likelihood of being uncovered or even possibly moved such that they could affect surrounding areas. The status of deeper sediments (Step 7, Decision 6) should be considered as data become available.

Rationale: Make use of historic information to appropriately guide subsequent sampling and analyses (which will almost always be required), and to avoid generating new data where data already exist.

3.2.1.2 Step 2: Develop and implement sampling plan

Based on Step 1 above, develop a Sampling and Analysis Plan (SAP – see Section 4.7) for review and approval by stakeholders, then implement same at both exposed and reference sites. The objective of the SAP is to fill in data gaps related to both COPCs and ROPCs. The SAP should not necessarily be restricted to surficial sediments. A determination is required as to whether there are any COPCs in the sediments that could be toxic and/or biomagnify up food chains (increase in concentrations through three or more trophic levels).

Some common contaminants that biomagnify

- Mercury¹
- PCBs^{2,3}
- DDT⁴
- Dioxins and furans³

1. Measure both total and methyl mercury concentrations in sediments (mercury only biomagnifies in the methylated form)
2. Measure total PCBs (sum of seven Aroclors: 1016, 1221, 1232, 1242, 1248, 1254, 1260; or the sum of congeners) as the PSQG are typically based on total PCBs or specific Aroclors.
3. If a detailed quantitative assessment is conducted, congener specific information may be required for sediments contaminated with PCBs, dioxins and/or furans to evaluate dioxin-like PCB congeners, dioxins and furans when converted into equivalent concentrations of 2,3,7,8-Tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD)
4. DDT breakdown products, DDD and DDE, should also be measured.

Decision Point 1: Two questions now need to be addressed. First, are COPC present in sediments above levels that have been shown to have minimal effects to biota living in the sediments? In other words, could the COPC possibly cause toxic effects? Typically only chemistry data will be available to characterize a site. These data are used in an initial pre-screening step to remove sites from further consideration if concentrations are below appropriate sediment toxicity thresholds. However, occasionally, biomonitoring data may be available for a site that indicates potential adverse effects are occurring. In this situation, the biomonitoring data are sufficient to suggest that additional assessment is needed regardless of the results of the screening step based on chemistry data alone. Second, do COPC present in sediments comprise substances that could biomagnify and affect the health of biological communities at higher trophic levels or of humans consuming biota contaminated with those substances?

The first question is addressed by comparing COPCs to the PSQG-LEL or other appropriate criterion (e.g., an SQG that predicts toxicity to less than 5% of the sediment-dwelling fauna, such as the Canadian Threshold Effect Level (TEL), or to a SQG-low from another jurisdiction). If no SQG exists, compare the COPC concentrations to those of the reference areas; sediments where concentrations are greater than 20% of the concentrations in the reference areas, and are statistically higher than the reference areas, suggest anthropogenic exposure has occurred. These substances should be considered as having the potential to cause toxic effects or biomagnify, and further assessment of the sediment is required. The second question is addressed by determining whether substances that can biomagnify are present at quantifiable concentrations. Two decisions are possible:

Comparison	Decision
All sediment COPC < SQG-low, <u>and</u> no substances present that can biomagnify	No further assessment or remediation required. <u>STOP</u>
One or more sediment COPC > SQG-low, <u>and/or</u> one or more substances present that can biomagnify	Potential risk; further assessment required. <u>PROCEED TO STEP 3</u>

Rationale: Conduct initial analyses as necessary to make a decision as to whether or not the sediments may pose a potential risk to the environment and/or to human health. By design, SQGs are typically conservative, in other words, over-protective. Thus, if sediment COPC concentrations are below SQG that predict minimal effects (SQG-low), there is negligible ecological risk. For example, Porebski et al. (1999) found that such SQG performed well as “levels below which unacceptable biological effects were unlikely to occur.” Because SQGs have no role in evaluating human health risks or biomagnification (Wenning and Ingersoll, 2002), and there are no such sediment guidelines, initial (conservative) decisions regarding biomagnification potential are simply based on the presence or absence of quantifiable amounts of substances that may biomagnify.

3.2.1.3 Step 3: Compare to reference (Is there a potential risk based on contaminant concentrations?)

Determine whether the concentrations of COPC exceeding SQG-low and/or concentrations of substances that can biomagnify statistically exceed reference concentrations as determined from reference area comparisons.

Decision Point 2: Two separate questions need to be addressed. First, are concentrations of COPC in sediments that are above SQG-low levels statistically different ($p < 0.05$) than reference conditions? Second, are concentrations of COPC that could biomagnify, which are present in sediments at quantifiable levels, not statistically different ($p < 0.05$) than those same COPC in reference areas? Note that in cases where there is little discriminatory power in statistical significance determinations due to very low variability in the reference areas (i.e., a very small difference from reference would be statistically significant but of arguable environmental significance), an additional comparison is possible, specifically: are concentrations of COPC less than 20% above those same COPC in reference areas? The +20% comparison is a straight arithmetic comparison of either mean or individual values, depending on site-specific circumstances ($\alpha = 0.05$; $\beta = 0.10$). Reference conditions include background conditions – either measured or determined from historical data. *Note, in making these comparisons, the data for an immensely contaminated (e.g., > 10 fold the SQGs that*

predict likelihood of toxicity), but relatively small area, should not necessarily be diluted with data from other, much less contaminated areas.

Comparison	Decision
[Concentrations of all sediment COPC > SQG-low and substances present that can biomagnify] < reference conditions and statistically no different than reference	No further assessment or remediation required. <u>STOP</u>
[Concentrations of one or more sediment COPC > SQG-low and/or one or more substances present that can biomagnify] > reference conditions and statistically higher than reference	Potential risk; further assessment required. <u>PROCEED TO STEP 4A</u>

Rationale: In this step, the framework is considering two possibilities: (1) Either all COPCs which are greater than SQG-low, and those which can biomagnify, are lower than reference (in this case there is no action required because sediment quality reflects background conditions) or (2) there is a difference from reference between one or more COPCs (which exceed SQG-low) and/or there is a difference from reference between one of more substances that can biomagnify. Inorganic and some organic substances occur naturally and may be naturally enriched in some areas (e.g., naturally mineralized areas, oil seeps). The focus of remediation efforts needs to be on anthropogenic (human) contamination, not natural enrichment. The additional possible determination of a difference of 20% between two sets of chemistry data is well within the bounds of typical analytical variability, may not represent a true (significant) difference because it is likely a consequence of natural sediment heterogeneity (MOE, 1996), and is highly unlikely to be of any environmental concern. The additional use of reference + 20% could be useful to screen out areas of marginal environmental concern, and is the same criterion as used for sediment toxicity test results comparisons (Section 3.2.2.2).

3.2.2 Preliminary quantitative assessment

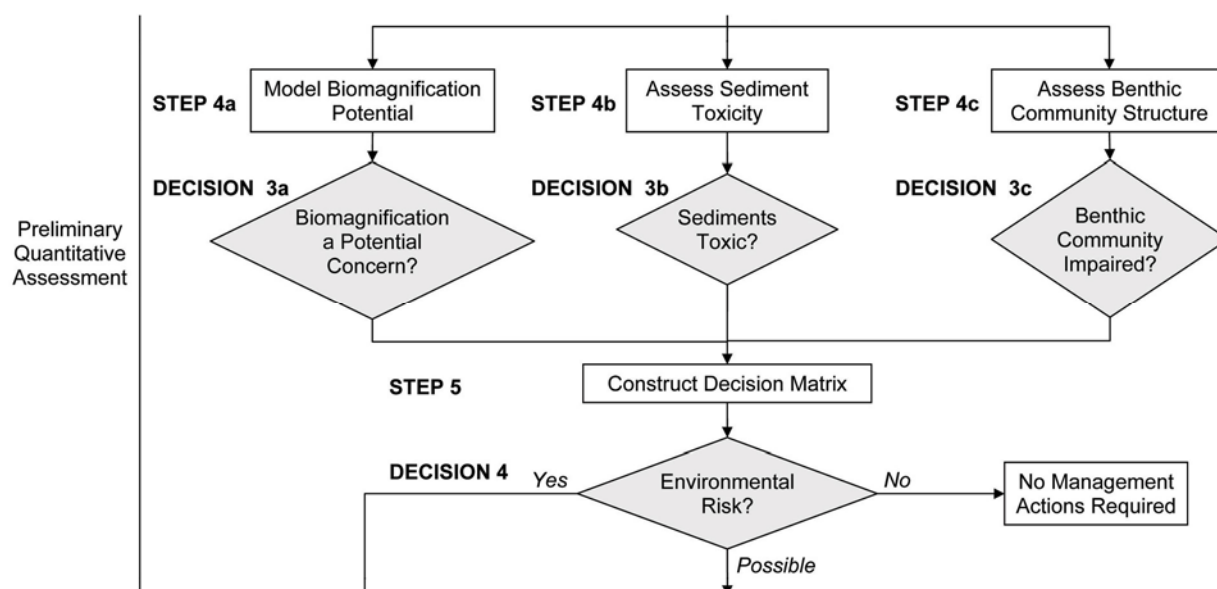


Figure 3. Preliminary Quantitative Assessment (Steps 4-5, Decisions 3-4). Contaminated areas screened in are further investigated, preparatory to determining whether there is or is not a problem, or whether additional investigations are required.

3.2.2.1 *Step 4a: Biomagnification potential*

If substances that can biomagnify remain of concern in sediment-dwelling organisms and in predators of those organisms through to top predators, concentrations in the sediments should be conservatively modeled to determine whether or not there is a potential risk (Grapentine et al., 2003a,b – See Section 5.2). Conservative modelling includes, for example: the assumption that maximum contaminant concentrations occur throughout the exposed area; the use of maximum biomagnification factors (BMFs); the assumption that fish feeding is limited to the exposure area. Basically, worst case scenarios, some of which may be unrealistic, are used to allow environmental risks to be either screened out or identified as possibilities to be investigated further.

Decision Point 3a: Determine whether or not contaminant biomagnification is a potential concern.

Comparison	Decision
There is no potential for contaminant biomagnification from the sediments through aquatic food chains	No further assessment or remediation required relative to biomagnification. <u>PROCEED TO STEP 4B</u>
There is potential for contaminant biomagnification from the sediments through aquatic food chains	Potential risk; further assessment of biomagnification potential required. <u>PROCEED TO STEP 4B</u>

Rationale: Conservative assumptions inherent in such a modeling exercise (i.e., worst case assumptions) will allow a determination either that biomagnification is not a concern, or that it may be a concern. In the latter case, additional site-specific assessment may be required (Step 6).

3.2.2.2 Step 4b: Sediment toxicity

For the remaining COPC, the PSQG LEL (SQG-low) and PSQG SEL (SQG-high) or equivalent should be used to map spatial patterns of contamination. For situations where no PSQG SEL exists, compare COPC concentrations to the Canadian Probable Effects Level (PEL) or to a SQG-high from another jurisdiction. The SQG-high should predict toxicity to 50% or more of the sediment infauna. Determine the toxicity of representative areas including those most heavily contaminated as well as those moderately and minimally contaminated, and reference areas, synoptic with sediment chemistry determinations (i.e., use subsamples of the same sample for both chemical analyses and toxicity testing). For situations where COPCs are greater than SQG-low but substantially less than SQG-high, best professional judgement should be used to determine if subsequent toxicity testing or bioassessment is required. Typically, laboratory sediment toxicity tests are conducted with three or four appropriately sensitive, standardized sediment-dwelling and/or sediment associated test organisms (e.g., *Hexagenia*, *Hyalella*, chironomids, oligochaetes) that are reasonably similar to those found (or expected to be found) at the site (based on available data – Step 1), and combined end-points that involve survival, growth and reproduction (i.e., acute and chronic endpoints); recognised methodologies and organisms (e.g. Bedard *et al.* 1992; Environment Canada 1997a,b; ASTM 2000) should be used in the assessment of sediment toxicity.

Decision Point 3b: Bulk sediment chemical analyses do not consider contaminant bioavailability, nor do they provide reliable information on the toxicity of sediment contaminants (reasonably reliable information can be obtained on the non-toxicity of sediment contaminants, cf. Decision Point 1). Thus, a determination is required as to

whether or not the sediments that were previously assessed as contaminated are toxic to individual organisms, and the extent of any toxicity.

Comparison	Decision
All sediment toxicity endpoints < 20% difference from reference and not statistically significantly different than reference	No further assessment required relative to laboratory toxicity. <u>PROCEED TO STEP 4C</u>
One or more sediment endpoints > 20% difference from reference and statistically significantly different than reference	Potential risk; further assessment required. <u>PROCEED TO STEP 4C</u>

Rationale: Although sediment toxicity tests have good power to detect differences between responses, a difference of 20% between controls and test/reference sediments is neither different nor environmentally relevant in short-term (e.g., 10 day), acute tests (Mearns et al., 1986; Suter, 1996; EPA/USACOE, 1998; Environment Canada, 1998, 1999). For this framework, sediments with less than a 20% difference between controls and test/reference sediments are not considered to be toxic, even if the difference is statistically significant.

3.2.2.3 Step 4c: Benthic community structure

Determine whether the benthic community is significantly different from appropriate reference sites. Two questions need to be addressed. First, is it appropriate or realistic to assess the benthic community? There may be situations where benthic community structure assessments relative to possible sediment contaminant effects are not appropriate or realistically possible (e.g., shallow harbours where propeller scour, dredging or other habitat disturbances alter benthic communities independent of any contaminant effects; dynamic sediment bedflow that may alter the biological zone as a result of deposition or scour). Benthic community structure assessments will also not be possible for sediments deeper than about 10 cm because the vast majority of the sediment-dwelling organisms live in shallower depths than 10 cm although some organisms (e.g., some bivalves) can burrow much deeper. Second, is the benthic community at the site significantly different from the benthic community in reference areas? Benthic community structure is often described in terms of the diversity, abundance, and dominance of different invertebrate species living in or on the sediment. Assessment of benthic community could include multimetric and/or multivariate analysis (as appropriate) to properly characterize the benthic community. Data interpretation using multivariate approaches are strongly recommended; however, the use of other metrics may have merit (Reynoldson et al, 1995, Hawkins et al, 2000, Barbour et al., 1999, Bailey et al., 2004, Environment Canada 2002).

Decision Point 3c: Determine benthic community impairment.

Comparison	Decision
It is inappropriate to assess the benthic community.	<u>PROCEED TO STEP 5</u>
Benthic community is not significantly different from reference areas.	<u>PROCEED TO STEP 5</u>
Benthic community is significantly different from reference areas.	<u>PROCEED TO STEP 5</u>

Rationale: Assessing the benthic community at a site, and comparing results to the community at appropriate reference areas, provides valuable information on the cumulative effect of multiple stressors on the invertebrate species that live in or on the sediment. Typically, benthic organisms reside at a site over most of their life span, and therefore integrate the effects of exposure to COPCs as well as other biological and physical stressors. Alteration in the benthic community may be related to the presence of elevated substances in the sediment but may also be due to other factors either natural (e.g., competition/predation, habitat differences) or human-related (e.g., water column contamination). A properly conducted field study and selection of appropriate reference sites are crucial for accurately assessing potential adverse effects to the benthic community at the site.

3.2.2.4 Step 5: Decision matrix

Develop a decision matrix based on, and ranking data from, the available Lines of Evidence (LOE): sediment chemistry, toxicity, benthos [if available and appropriate] and biomagnification potential (Table 5: adapted from Grapentine et al., 2002a). Samples for sediment chemistry and toxicity should be collected synoptically (sub-samples of the same samples); samples for benthos are collected coincidentally (i.e., at the same locations but not on the same samples). Samples for benthos and chemistry analyses can be collected during initial field sampling and archived until and unless needed, thus reducing field costs. However, samples for sediment toxicity cannot be archived for longer than 8 weeks and should ideally be tested as soon as possible following collection (EPA/USACOE, 1998). If benthos studies are not reasonably possible, fit other LOE into Table 6 and use best professional judgement in Step 6.

Decision Point 4: At this point a definitive decision may be possible. Specifically, sufficient information has now been gathered to allow for an assessment of three possibilities: (1) the contaminated sediments pose an environmental risk (see Section 9: Risk Management); (2) the contaminated sediments may pose an environmental risk,

but further assessment is required before a definitive decision can be made; (3) the contaminated sediments pose a negligible environmental risk. In Table 6, definitive determinations are possible in four of the 16 possible scenarios (two determinations of negligible environmental risk requiring no further actions; two of environmental risk requiring management actions).

Rationale: At this point definitive determinations are possible in some cases with the provision that sediment stability may still need to be assessed (Step 7); in other cases, further assessment is needed, but can be guided by the results of this data integration. As noted by Wong (2004), SQGs do not provide definitive information for decisions regarding contaminated sediments, including remediation; a weight of evidence (WOE) approach is required. In a WOE approach, sediment chemistry data are given the least weight (Section 3.1, “rules” 1 and 2); benthic community data are given the most weight (Section 3.1, “rule” 3).

The type of WOE integration of LOE shown in Table 7 is usually applied on a station-by-station basis. Thus, although initial screening (Steps 1-3) is intended to screen out areas with relatively low contaminant concentrations, subsequent more detailed sampling of these areas may include stations with contaminant concentrations below levels of concern. Mapping of the results is one means to apply the findings on a large sample basis (i.e., to all sample locations), as a tool for expert/stakeholder groups to identify and focus on obvious problem areas/patterns.

Table 5. Ordinal ranking for weight-of-evidence categorizations for chemistry, toxicity, benthos and biomagnification potential.

	●	◐	○
Bulk Chemistry (compared to SQG)	Adverse Effects Likely: One or more exceedances of SQG- high	Adverse Effects May or May not Occur: One or more exceedances of SQG- low	Adverse Effects Unlikely: All contaminant concentrations below SQG-low
Toxicity Endpoints (relative to reference)	Major: Statistically significant reduction of more than 50% in one or more toxicological endpoints	Minor: Statistically significant reduction of more than 20% in one or more toxicological endpoints	Negligible: Reduction of 20% or less in all toxicological endpoints
Overall Toxicity	Significant: Multiple tests/endpoints exhibit major toxicological effects	Potential: Multiple tests/endpoints exhibit minor toxicological effects and/or one test/endpoint exhibits major effect	Negligible: Minor toxicological effects observed in no more than one endpoint
Benthos Alteration (multivariate assessment, e.g., ordination)	“different” or “very different” from reference stations	“possibly different” from reference stations	“equivalent” to reference stations
Biomagnification Potential (relative to reference)	Significant: Based on Step 6	Possible: Based on Step 4a	Negligible: Based on Steps 4a or 6
Overall WOE assessment	Significant adverse effects: elevated chemistry; greater than a 50% reduction in one or more toxicological endpoints; benthic community structure different (from reference) ; and/or significant potential for biomagnification	Potential adverse effects: elevated chemistry; greater than a 20% reduction in two or more toxicological endpoints; benthic community structure possibly different (from reference); and/or possible biomagnification potential	No significant adverse effects: minor reduction in no more than one toxicological endpoint; benthic community structure not different from reference; <u>and</u> negligible biomagnification potential

SQG = Sediment Quality Guideline; EC = Effective Concentration. Note that the overall definition of “No Significant Adverse Effects” is independent of sediment chemistry.

Table 6. Decision Matrix for WOE Categorization. Based on Table 5, see text for explanation; a dash means “or”. Separate endpoints can be included within each LOE (e.g., metals, PAHs, PCBs for Chemistry; survival, growth, reproduction for Toxicity; abundance, diversity, dominance for Benthos).

SCENARIO	BULK SEDIMENT CHEMISTRY	OVERALL TOXICITY ¹	BENTHOS ALTERATION ²	BIOMAGNIFICATION POTENTIAL ³	ASSESSMENT
1	○	○	○	○	No further actions needed
2	●-○	○	○	○	No further actions needed
3	○	○	●-○	○	Determine reason(s) for benthos alteration (Section 7.3)
4	○	●-○	○	○	Determine reason(s) for sediment toxicity (Section 7.3)
5	○	○	○	●	Fully assess risk of biomagnification (Section 6.3)
6	●-○	●-○	○	○	Determine reason(s) for sediment toxicity (Section 7.3)
7	○	○	●-○	●	Determine reason(s) for benthos alteration (Section 7.3) and fully assess risk of biomagnification (Section 6.3)
8	●-○	○	●-○	○	Determine reason(s) for benthos alteration (Section 7.3)
9	●-○	○	○	●	Fully assess risk of biomagnification (Section 6.3)
10	●-○	●-○	○	●	Determine reason(s) for sediment toxicity (Section 7.3) and fully assess risk of biomagnification (Section 6.3)
11	●-○	○	●-○	●	Determine reason(s) for benthos alteration (Section 7.3) and fully assess risk of biomagnification (Section 6.3)
12	○	●-○	○	●	Determine reason(s) for sediment toxicity (Section 7.3) and fully assess risk of biomagnification (Section 6.3)
13	○	●-○	●-○	○	Determine reason(s) for sediment toxicity and benthos alteration (Section 7.3)
14	○	●-○	●-○	●	Determine reason(s) for sediment toxicity and benthos alteration (Section 7.3), and fully assess risk of biomagnification (Section 6.3)
15	●-○	●-○	●-○	○	Management actions required ⁴
16	●-○	●-○	●-○	●	Management actions required ⁴

¹ Overall toxicity refers to the results of laboratory sediment toxicity tests conducted with a range of test organisms and toxicity endpoints. A positive finding of sediment toxicity may suggest that elevated concentrations of COPCs are adversely affecting test organisms. However, toxicity may also occur that is not related to sediment contamination as a result of laboratory error, problems with the testing protocol, or with the test organisms used.

² Benthos alteration may be due to other factors, either natural (e.g., competition/predation, habitat differences) or human-related (e.g., water column contamination). Benthos alteration may also be related to sediment toxicity if a

substance is present that was not measured in the sediment or for which no sediment quality guidelines exist, or due to toxicity associated with the combined exposure to multiple substances.

³ Per Table 5, significant biomagnification (●) can typically only be determined in Step 6; Step 3 only allows a determination that there either is negligible biomagnification potential or that there is possible biomagnification potential. However, there may be site-specific situations where sufficient evidence is already available from fish advisories and prior research to consider biomagnification at a site significant; this would be determined in Step 1 (examination of available data). Thus, for example, if significant biomagnification were indicated in Scenario 5, above, management actions would be required. The other three LOE do allow for definitive determinations in prior Steps of this framework.

⁴ Definitive determination possible. Ideally elevated chemistry should be shown to in fact be linked to observed biological effects (i.e., is causal), to ensure management actions address the problem(s). For example, there is no point in removing contaminated sediment if the source of contamination has not been addressed. Ensuring causality may require additional investigations such as toxicity identification evaluation (TIE) and/or contaminant body residue (CBR) analyses (see Section 6.3). If bulk sediment chemistry, toxicity and benthos alteration all indicate that adverse effects are occurring, further assessments of biomagnification should await management actions dealing with the clearly identified problem of contaminated and toxic sediments adversely affecting the organisms living in those sediments. In other words, deal with the obvious problem, which may obviate the possible problem (e.g., dredging to deal with unacceptable contaminant-induced alterations to the benthos will effectively also address possible biomagnification issues).

3.2.3 Detailed quantitative assessment

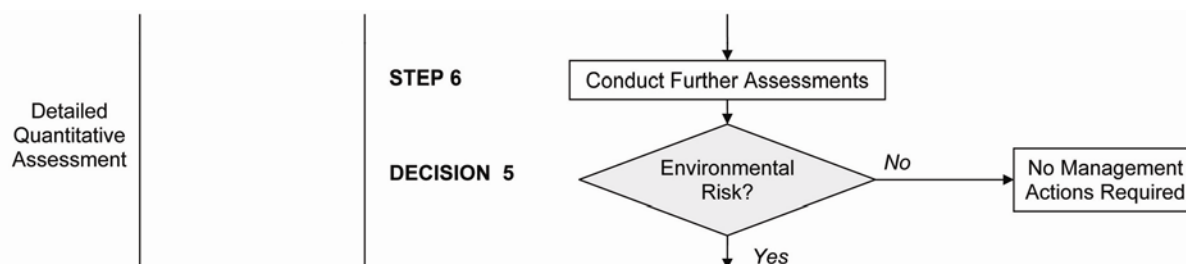


Figure 4. Detailed Quantitative Assessment (Step 6, Decision 5). Decisions can be made regarding management actions for specific situations. In other situations, additional, focused investigations will be required.

3.2.3.1 Step 6: Further assessment

As per the 16 possible scenarios in Table 6, four result in definite decisions and 12 scenarios result in a determination that the contaminated sediments may pose an environmental risk, but further assessment, outlined in Table 6, is required before a definitive decision is made.

Decision Point 5: Based on additional investigation, determine whether or not an environmental risk exists. *This is where, in particular, and as noted in Section 3.2., it is critical that the study team include scientists with strong expertise in sediment chemistry (chemical fate, transport and speciation), sediment toxicity testing, benthic community*

assessment, food chain effects and environmental statistics for the design, implementation, and interpretation of both the previous and any additional investigative studies required.

Rationale: (1) If there is no clear link between elevated chemistry (i.e., sediment contaminant concentrations > SQG-low) and biological effects (i.e., sediment toxicity and/or benthos alteration), there may be no point to sediment remediation as, if the sediment contaminants are not causative, sediment remediation will not ameliorate the biological effects. It is necessary to conduct more detailed studies to determine the cause of biological effects. (2) Observed toxicity and/or benthos alteration in the absence of elevated chemistry may be due to unmeasured contaminants or non-contaminant-related factors; either way, certainty as to causation is required (e.g., toxicity identification evaluation, TIE). (3) Modeling biomagnification only indicates whether there is no problem or may be a problem; if there is a potential biomagnification problem, more definitive assessments involving field measurements (e.g., contaminant body residue [CBR] analyses), laboratory studies, and/or more realistic modeling scenarios are required (see Section 6.3).

The previous assessments typically focus on surficial sediments (up to about 10 cm depth). Surficial sediments effectively cover deeper sediments, which may be similarly or differently contaminated. If so, there is a need to determine whether, under unusual but possible natural or human-related circumstances, these deeper sediments may be uncovered. Such studies involve an assessment of both sediment stability and sediment deposition rates.

3.2.3.2 Step 7: Deeper sediments

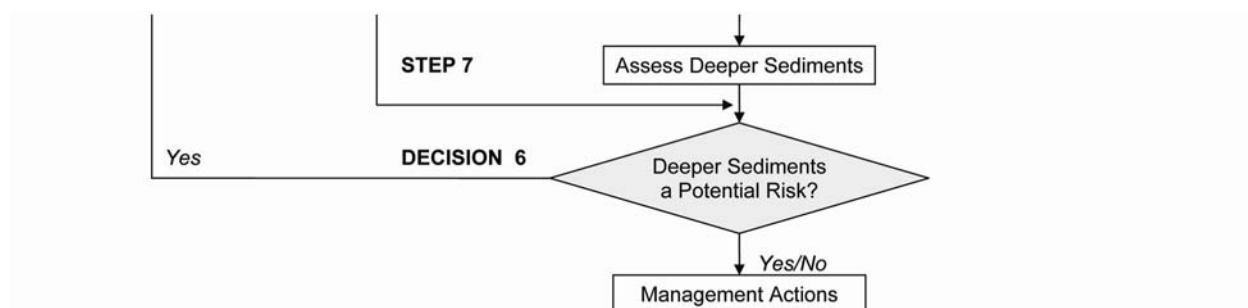


Figure 5. Assessment of Deeper (Below Surficial) Sediments (Step 7, Decision 6). If deeper sediments may pose a risk and could be exposed, the risk posed and need for management actions need to be determined.

Decision Point 6:

Comparison	Decision
Levels of COPC in deeper sediments below SQG-low and no substances present that can biomagnify, <u>or</u> deeper sediments very unlikely to be uncovered under any reasonably possible set of circumstances	No further assessment or remediation required. <u>STOP.</u> Management options for polluted surficial sediments should be determined.
Levels of COPC in deeper sediments above SQG-low and/or one or more substances present that can biomagnify, <u>and</u> these sediments may be uncovered under one or more reasonably possible set of circumstances	Potential risk; further assessment may be required (See Guidance, Section 1, “rule” 1). <u>FOLLOW THE FRAMEWORK FROM STEP 1 (IF NECESSARY).</u> Necessary information will probably already have been gathered for some initial steps.

Rationale: If deeper sediments are contaminated, and could be uncovered, they could pose an environmental risk, which needs to be evaluated. If the sediments are not likely to be uncovered, i.e., to become surface sediments, under any reasonably likely set of circumstances (e.g., a 100-year flood), then they do not require further assessment as any contaminants they contain will remain buried and there will be no exposure routes to biota.

4 ERA components of the framework: problem definition (screening assessment)

The following four sections of this document (section 4, 5, 6, and 7) provide information regarding key components of the ecological risk assessment (ERA) approach upon which the decision-making framework is explicitly based. The information provided is not, and is not intended to be, exhaustive (i.e., this document is not a “cook book”); rather, it is intended to provide readily understandable supporting information.

A Screening Assessment (Figure 2, Sections 3.2.1 to 3.2.3) involves simple, qualitative and/or comparative methods, with heavy reliance on literature information and previously collected data (CCME, 1996). Uncertainty (Section 7.4) is highest at this level of an ERA due to the use of conservative methodology and assumptions. Screening on

both a conservative and a less conservative basis can provide a range of possible outcomes (which thus need to be investigated). Note that there is no single correct way to conduct this or other levels of an ERA. Subsequent ERA levels or tiers are conducted in an iterative approach, which generally means testing of hypotheses and conclusions and re-evaluating assumptions as new information is gathered.

4.1 Site definition

Sites typically comprise samples from multiple stations, and can be delineated based on ecologically defined scales (Section 7.1), on contaminant concentrations, or on other site-specific conditions. Within such delineations, species at risk and their habitats need to be considered, including the minimum home range of fish feeding on benthic invertebrates. Two additional determinations are needed: (1) does the site have a high level of environmental sensitivity based on habitat (not land use), e.g., is it a wetland used by migrating waterfowl or a feeding ground for shellfish or fish; (2) is it contaminated only from off-site sources, which themselves need to be evaluated? These determinations will affect the design and implementation of subsequent investigations.

Further, the energy of the aquatic system should be considered in determining site boundaries. In a high energy system, sediments may be washed downstream and deposited distal to the site. Likewise, evaluations of scour and deposition may show that sediments at depth may or may not be of concern or that the study area is potentially impacted from upstream sites.

4.2 Contaminants of potential concern (COPCs)

Two classes of COPCs need to be considered:

1. Contaminants that can cause acute (short-term, e.g., death) or chronic (longer-term, e.g., effects on growth and/or reproduction) effects to biota. The potential risk from these contaminants is assessed based on comparisons to SQG-low. Where SQG-low are not available for particular contaminants, it may be possible to derive similar values using numerical methods from compilations of toxicity test data, such as species sensitivity distributions (SSDs). Note that SQGs of any sort are, by definition, preliminary, due to data limitations (O'Connor, 2004).
2. Contaminants that can biomagnify up food chains. Biomagnification is restricted to organic substances, e.g.: methyl Hg; PCBs; DDT; 2,3,7,8-TCDD.

4.3 Receptors of potential concern (ROPCs)

Primary receptor species must both be potentially exposed to sediment contaminants (the COPCs), and be relevant to the area being assessed (i.e., live or be expected to

live primarily in that area). Secondary receptor species are the consumers of the primary receptor species. If it is not appropriate to follow the MOE protocol (Bedard *et al.* 1992) or other standard toxicity test methods (e.g. Environment Canada 1997a,b; ASTM 2000), agreement among stakeholders is required *a priori* regarding which receptor species to use for assessments and what surrogate species (if necessary) to use for toxicity testing.

4.4 Assessment of endpoints and measures of effects

An assessment endpoint is defined as the explicit expression of the environmental value that is to be protected. Examples of assessment endpoints include survival, growth and reproduction of major aquatic communities (e.g., aquatic plants, benthic invertebrates (bottom-dwelling animals without backbones), fish, aquatic-dependent birds and mammals). Generic ERA assessment endpoints are provided by the USEPA (2003). A measure of effect is defined as the measurable ecological characteristic that is related to the assessment endpoint. Measures of effect comprise the actual measurements (e.g., actual determinations of survival, growth and reproduction via laboratory or other tests and/or field observations).

4.5 Reference areas/ locations

Reference areas/locations serve as the benchmarks against which to compare the contaminated sites. Typically, reference areas/locations represent “the optimal range of minimally impaired conditions that can be achieved at sites anticipated to be ecologically similar” and should be acceptable by local stakeholders and appropriately represent reference conditions (Krantzberg *et al.*, 2000). Ideally the same number of reference sites would be assessed as exposed sites; realistically, a smaller number can be used provided reference conditions are adequately quantified. However, some study areas may provide few or no suitable reference sites, and would be better sampled with a gradient array of sites.

Environment Canada has developed reference conditions for Great Lakes sediments based on a large data set of stations for three groups of parameters: physico-chemical attributes; toxicity; and, benthic community structure. Thus, exposed areas/locations can be compared to appropriate reference conditions by a variety of statistical methodologies (Reynoldson and Day, 1998; Reynoldson *et al.*, 2002a).

Reference areas/locations can be used for three main applications (Apitz *et al.*, 2002): to determine whether or not a contaminated area may require remediation; to determine incremental risk (between an exposed and reference site); and, in a post-remedial monitoring program.

4.6 Conceptual site model (CSM)

The conceptual site model (CSM) is a critical component of any sediment (or other) ERA assessment. It should involve both temporal and spatial components and be reviewed by regulatory agencies and other stakeholders prior to commencing field or laboratory studies to ensure there is agreement. It comprises *“a three-dimensional description of a site and its environment that represents what is known (or suspected) about the contaminant source area(s), as well as, the physical, chemical, and biological processes that affect contaminant transport from the source(s) through site environmental media to potential environmental receptors. The CSM identifies assumptions used in site characterization, documents the relevant exposure pathways at the site, provides a template to conduct the exposure pathway evaluation and identifies relevant receptors and endpoints for evaluation. CSM development is an on-going, iterative process that should be initiated as early as possible in the investigative process. The CSM should be as simple or as complex as required to meet site objective(s). The CSM is also an important communication tool to facilitate the decision-making processes at the site”* (Apitz et al., 2002). Work done at similar sites can assist in identifying potential shortcomings and pitfalls, and help focus the CSM to the extent possible.

4.7 Sampling and analysis plan (SAP)

The Sampling and Analysis Plan (SAP) is developed based on all of the previous considerations (Sections 4.1 to 4.6). Its initial goal is to identify potential contaminant sources and to delineate areas of contamination (their full nature and their spatial – vertical and lateral – distribution) for subsequent investigation. Subsequent goals involve other LOE as per Figure 1. If a detailed quantitative assessment is conducted, congener specific information may be required for sediments contaminated with PCBs, dioxins and/or furans to evaluate dioxin-like PCB congeners, dioxins and furans when converted into equivalent concentrations of 2,3,7,8-Tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD).

5 ERA components of the framework: exposure assessment

The decision-making framework is specific for environmental concerns associated with contaminated sediment, including not only ecological, but also human health concerns related to biomagnification. However, there may be situations where potential human health concerns are associated with dermal contact to contaminated sediment (e.g., swimming, wading), or by other exposure routes (e.g., flooding resulting in aquatic sediments contaminating residential soils or gardens, unacceptably high levels of contaminants that do not biomagnify such as Cd, Pb, PAHs in shellfish or fish). In such situations, a screening level HHRA should be considered to assess potential risks and inform the public.

5.1 Sediment chemistry – preliminary quantitative

Preliminary quantitative assessment of sediment contaminants (Figure 2, Sections 3.2.1.1 to 3.2.1.3) can be done on the basis of individual contaminants or by using specific groups of contaminants as surrogates (Grapentine et al., 2002b). Combining information on different contaminants (e.g., Marvin et al., 2004) is not recommended due to information loss. However, where the mode of action and target effect of a toxicant are the same, additivity of contaminants can be considered. In some circumstances, an examination of integrated information from several types of contaminants (i.e., use of a Sediment Quality Index) could contribute to the overall interpretation of the data. Relying solely on such integrated information is not advised. Ancillary information required includes, but is not limited to, sediment particle size and total organic carbon (TOC) data. The extent of contamination can be characterized using techniques such as grids, random and stratified random sampling; the decision regarding which particular method to use will be site-specific.

5.2 Biomagnification potential – preliminary quantitative

Uptake, bioaccumulation and biomagnification of chemicals through the food chain, which is restricted to very few organic chemicals (e.g., methyl mercury; DDT; PCBs; 2,3,7,8-TCDD) should be considered on a case-by-case basis (Figure 3, Section 3.2.2.1). Fish advisories can provide useful information regarding issues (chemicals and species) related to biomagnification. Guidance in initial modeling efforts is provided in Grapentine et al. (2003a,b). Essentially, *“this approach relies on the application of conservative (i.e., protective) assumptions regarding BMFs and tissue residue criteria (TRC) to screen for potential toxicological effects to receptor species at higher trophic levels as the result of biomagnification from benthic invertebrate tissue through the food web”* (Duncan Boyd, pers. comm.). Benthic invertebrate tissue concentrations are used to predict concentrations in higher trophic levels.

5.3 Detailed quantitative

Detailed quantitative assessment within the framework is outlined in Figure 4 (Sections 3.2.3.1 and 3.2.3.2). Because fish are mobile, their entire feeding area needs to be considered in order to fully assess the potential for some organic contaminants to biomagnify (e.g., through area curve modeling - Freshman and Menzie, 1996). Factors such as site- and species-specific BMFs, lipid content, age/size, and receptor food preference can also be incorporated. Utilizing more realistic assumptions than those used for preliminary quantitative assessment should allow for a better determination regarding the toxicological outcome for upper trophic level receptor species. Whereas the preliminary quantitative assessment is solely a modeling exercise based on sediment and benthos, this more detailed quantitative assessment involves other food chain measurements including fish and possibly plankton.

Natural fate and transport processes affecting sediment contaminants must also be considered, and could include: in-bed fate processes, including irreversible adsorption and chemical or biological reactions; in-bed transport processes, including diffusion and advection; interfacial transport processes, including sediment deposition and resuspension and bioturbation. Potential contaminant sources from groundwater should also be considered. Direct field evidence will be required in some cases. In other cases, reasonable assumptions may be possible based on scientific knowledge and best professional judgement.

More detailed sediment chemistry exposure assessment related to determination of causation could, in some cases, involve the use of biomarkers. Multiple biomarkers can be used in their own WOE assessment as part of the overall ERA (Galloway et al., 2004).

6 ERA components of the framework: effects assessment

6.1 Toxicity testing – preliminary quantitative

The magnitude of any toxicity (Figure 3, Section 3.2.2.2) associated with exposure to contaminants in the sediments is assessed. Such information is typically determined from sediment toxicity tests with well-established, standard test organisms (e.g. Bedard *et al.* 1992; Environment Canada 1997a,b; ASTM 2000). The possibility of toxicity due to factors other than the COPCs (e.g., grain size, ammonia, sulphides) is typically considered as part of standardized test procedures. Various approaches are possible for integrating multiple toxicological endpoints into a single LOE, however the results of laboratory toxicity tests do not reliably predict effects to field populations (Suter, 1996; Reynoldson et al., 2002a; Chapman et al., 2002).

6.2 Benthos alteration – preliminary quantitative

Benthos alteration (Figure 3, Section 3.2.2.3) is assessed by identifying and enumerating benthic assemblages, and using both univariate (e.g., species richness, abundance, dominance) and multivariate analyses (e.g., ordination, principle component analysis [PCA]) to determine similarities and differences from reference areas and/or conditions (Chapman, 1996; Simpson et al., 2005).

6.3 Detailed quantitative

Detailed quantitative toxicity assessment (Figure 4, Table 6; Section 3.2.3.2) involves additional or more extensive studies as appropriate to site-specific circumstances, for example: spiked sediment toxicity tests; Toxicity Identification Evaluation (TIE); Critical Body Residue (CBR) analyses; tests with resident organisms; in situ bioassays.

Spiked sediment toxicity tests involve adding increasing concentrations of one or more suspected toxicants to reference sediment, and determining concentrations at which effects occur compared to exposed sediments. This procedure can also be applied to exposed sediments. It assists in identifying causative agents for observed toxicity and/or benthic community alterations. Similar information can be provided by TIE and CBR.

TIE were originally based on water or effluent toxicity tests and involve manipulating the chemical composition of toxic samples to remove specific substances (e.g., metals, ammonia) followed by retesting (Burgess, 2000). When an expected toxic effect is not observed as a result of removing specific substance(s), those substance(s) are added back, and the toxic effect is reassessed to confirm that those substances are indeed responsible for the initially observed toxicity, and that toxicity recurs at about the same levels as initially. TIE were subsequently applied to sediment pore waters (assuming that most of the toxicity observed in sediments was due to aqueous exposure routes) (Ankley and Schubauer-Berigan, 1995). They have recently been applied to whole sediments in the marine environment, and although procedures are not yet available to perform full TIE on whole sediments, those procedures that are available show good promise (Burgess et al., 2000; Pelletier et al., 2001; Burgess et al., 2003; Ho et al., 2004). A chemical fractionation scheme has been used together with toxicity testing, to attempt to determine causation in whole sediment freshwater toxicity tests in Lake Ontario (McCarthy et al., 2004).

CBR determinations are based on the fact that, for a contaminant to cause toxicity to an organism, that contaminant has to contact a biological receptor, which generally means the contaminant must be bioaccumulated (taken up) by the organism. Though this remains an active area of research, contaminant concentrations in organisms have been linked to effects (Jarvinen and Ankley, 1999), and used to determine causation in WOE determinations (e.g., Borgmann et al., 2001).

Testing the responses of resident organisms may be appropriate to determine, for instance, why laboratory tests with standard organisms indicate toxicity, but there are no alterations to resident benthic communities. It is entirely possible that resident organisms are more tolerant to sediment contaminants than laboratory organisms (Chapman et al., 2003). If tolerance has been established, then whether or not there are also costs in terms of the loss of intolerant species or energetic costs to the tolerant organisms should be determined.

In a similar manner, in situ bioassays (toxicity and/or bioaccumulation) can be used to test for differences between responses in the laboratory and in the field. Laboratory bioassays are conducted under controlled conditions that will not mimic field conditions to which resident populations are exposed. Conducting bioassays in situ and comparing the results to laboratory tests can assist in determining why differences in responses occur, and whether or not resident populations are at risk (laboratory bioassays tend to be conservative).

7 Risk characterization

The basic approach of starting with chemical hazard assessment (i.e., the use of SQGs: Figure 2), then adding toxicity tests, followed by receiving environment evaluations (Figure 3), matches current practices in the Great Lakes and other parts of Canada as well as the USA (Krantzberg et al., 2000), and international trends (Power and Boumphrey, 2004; Apitz et al., 2005). The framework, outlined here, can be applied to large and small sites in terms of both preliminary and more detailed assessments. It fits within the ERA paradigm, and provides information necessary for the protection of both local aquatic communities and endangered species. The framework also differentiates between those scenarios where elevated concentrations of contaminants are associated with adverse biological effects and those scenarios where they are not (since the presence of substances in sediments where they would not normally be found, or at concentrations above natural background levels, does not necessarily mean that adverse biological effects are occurring). The following documents provide additional detailed information regarding various LOE mentioned herein and their eventual use in risk characterization: MacDonald et al. (2002a,b); Ingersoll and MacDonald (2002); Suter et al. (2002).

7.1 Issues of scale

Issues of scale need to be considered on a site- and situation-specific basis, and are an important factor in choosing between management actions and further study. Estimated exposure from a large area is usually much lower than exposure from a specific, localized site. Under the Contaminated Sites process, the Ministry (MOE) does not allow the relatively high risks of small “hot spots” to be “averaged down” by the relatively small risks of the less contaminated surrounding area. Further, ERA should not be used to avoid addressing an extreme, local “hot spot”. However, considerations of biomagnification potential at a Detailed Quantitative level need to consider the feeding ranges (area use) and preferences of fish and waterfowl (i.e., the measured or assumed fraction of a predator’s diet that is represented by a particular prey species). Area use represents the proportion of a prey species’ home range associated with a particular area of contaminated sediments, and can include seasonal exposure during critical life stages or diminished exposure of migratory species.

7.2 Preliminary quantitative

A Preliminary Quantitative ERA (Tables 1 and 2, Section 3.2.3.1) provides more quantitative information than a Screening Assessment, reduces uncertainty, and is more extensive and expensive (CCME, 1996). Exposure and effects assessments are integrated to determine whether or not significant effects are occurring or are likely to occur. In addition, the nature, magnitude, and extent of effects on the selected assessment points are described. The substances that may be causing or substantially

contributing to such effects (the contaminants of potential concern: COPCs) are identified to the extent possible.

The results for each LOE are compiled and interpreted separately. Subsequently, they are combined and integrated, including uncertainty and best professional judgement, to establish a WOE for assessing risks (e.g., Chapman et al., 2002; Reynoldson et al., 2002b). WOE approaches need to be: as quantitative as possible; transparent; and, draw on a broad range of interdisciplinary expertise (Burton et al., 2002). Risks of adverse effects can generally be considered in four categories:

- Negligible – similar to those for reference conditions
- Moderate – minor or potential differences compared to reference conditions
- High – major or significant differences compared to reference conditions
- Uncertain – requiring further study (e.g., a Detailed Quantitative assessment).

7.3 Detailed quantitative

A detailed quantitative assessment (Table 6, Section 3.2.3.2) is the most extensive form of ERA, relying on site-specific data and predictive modeling; information is as quantitative as possible (CCME, 1996). It is intended to reduce key uncertainties in a transparent and scientifically sound manner such that final decisions can be made for all potential contaminated sediment scenarios. Typically, lower ERA tiers involve conservative or ‘worst case’ assumptions. This higher tier of ERA typically involves more realistic assumptions.

Detailed quantitative assessment also generally involves determination of causation, specifically answering the question as to whether or not any observed biological effects are due to sediment contaminants and, if so, which contaminant(s) and at what concentration(s) (e.g., Suter et al., 2002). Although sediment stability issues can be addressed initially in a Preliminary Quantitative ERA, they are conclusively addressed here.

Risks will generally be considered in three categories:

- Negligible – similar to those for reference conditions
- Moderate – minor or potential differences compared to reference conditions
- High – major or significant differences compared to reference conditions.

7.4 Uncertainty

Scientific investigations do not always result in easy answers. Uncertainty is inherent in any and all ERA. However, the ERA process is designed to accommodate the relationship between scientific uncertainty and the ability of risk managers to make risk management decisions. The goal in progressing from screening to more quantitative

assessment is to diminish key uncertainties and improve confidence in the decision-making process.

In the case of biomagnification assessments, site-specific data and locally relevant food-web structure will diminish the uncertainty associated with extrapolations from literature-based models. However, food-web modeling and predictions will still be required to evaluate possible effects related to biomagnification. Thus, uncertainty cannot be totally eliminated.

There are two general types of uncertainty. Stochastic uncertainty refers to the inherent randomness of the system being assessed, and can be described and estimated but cannot be reduced. Uncertainty arising from human error or from imperfect knowledge can, however, be reduced. In the case of biomagnification assessments, the major sources of the latter type of uncertainty are variability in model inputs (empirically observed variation and/or lack of data for key parameters, and the assumptions and simplifications which are inherent to the structure of any particular model).

Stochastic uncertainty results in intrinsic model limitations that are not the result of a lack of data or computational power. For example, food web model predictions are considered good if they are within a factor of five of observed concentrations for upper trophic level receptors. This leaves a considerable measure of uncertainty for decision-makers to deal with, since this margin of error will frequently exceed the scale of the relative improvement in ecosystem outcome which is desired.

CCME (1996) requires the identification of “key uncertainties”, a management decision as to whether they are acceptable or not, and an evaluation as to whether a preliminary quantitative ERA exposure assessment would significantly reduce uncertainty. The USEPA (1988) identifies the importance of quantitative uncertainty analysis and has published a policy for use of probabilistic analysis in risk assessment.

Three common methods for dealing with sources of uncertainty are sensitivity analysis, Monte Carlo simulation, and the use of monitoring data for model calibration. Sensitivity analysis is a fundamental requirement of any model application and geared to ensuring that the level of effort applied to improving the accuracy of model input parameters is commensurate with their effect on the accuracy of modeled output. Input parameters which have only a small effect on the accuracy of modeled output can be estimated by less accurate and costly methods. Once sensitivity analysis has identified the critical input parameters, a Monte Carlo analysis provides a stochastic approach to generating probabilistic model output through repetitive model runs using the distribution characteristics of uncertain model input parameters. The probability distributions associated with this approach provide an excellent means of quantifying model uncertainty. However, unless the input parameter distribution characteristics are derived from actual data, the uncertainty in outputs is purely a function of assumptions made about the uncertainty of input parameters. Model calibration using monitoring data is an obvious and necessary means of diminishing uncertainty, but good modeling practice

requires that model calibration and validation use independent data to avoid assuming that which is to be predicted.

Progression from a screening level assessment, to a more quantitative assessment incorporating site-specifically derived values such as biomagnification factors (BMF), area use factors, and food preference factors for receptor species may result in some reduction of uncertainty compared with the use of literature values. It may also improve the ability to quantify and partition uncertainty. However, the achievable reduction in uncertainty requires careful evaluation before the decision is made to proceed with a more quantitative risk assessment, since it may not diminish uncertainty to the point where decision-making becomes any more straightforward. If the analysis demonstrates that the potential for significant reduction in uncertainty is limited, then the risk manager must evaluate whether the benefits of the ensuing marginal decrease in uncertainty justify the corresponding time and costs. It may prove more expedient to proceed to an examination of risk management options, particularly in cases where socio-economic or technological constraints may limit these options.

In order to ensure that the allocation of time and resources to a quantitative ERA will sufficiently diminish uncertainty for risk management decision-makers, a quantitative uncertainty analysis must be applied at all sites as a prerequisite for proceeding from a screening level ERA to a quantitative ERA. This requirement is generic and not specific to biomagnification assessment.

In the specific case of biomagnification assessment, the accuracy of model predictions of tissue residues in third or fourth trophic level receptor species cannot be quantitatively validated using site-specific data due to the complexity of such food chain transfers, and hence site-specific tissue residue data should only be used to qualitatively ground-truth model predictions. Because sensitivity analysis will generally identify benthic invertebrate tissue concentrations as the most critical measurable input parameter in food chain models, measurement of invertebrate tissue residues should be used as the primary means of assessing biological exposure.

Section III: MANAGEMENT

Prior to any remedial action, permits and approvals may be required under different legislation. An overview of legislation which may impact sediment management decisions, as well as a brief overview of some remedial options, was previously outlined in *An Integrated Approach to the Evaluation and Management of Contaminated Sediments* (MOE 1996). This guidance has been updated here to reflect new and amended legislation. However, this document is not intended to provide legal advice and it should not be construed as such. The relevant and applicable legislation should be reviewed and considered before proceeding with a project. It may also be prudent to consult with a lawyer before proceeding with any sediment management option.

Updates and amendments to legislation can be accessed through the e-laws web site (www.e-laws.gov.on.ca).

8 Legislation

8.1 Legislative requirements

Prior to commencing any remediation project, consideration needs to be given to which permits and approvals will need to be obtained and from which appropriate agencies. Depending upon site-specific conditions, a project may require a number of permits or approvals from various government agencies and levels of government, all of which may have jurisdiction over certain aspects of the project.

For example, most large sediment removal projects undertaken by the Province will be subject to the requirements of the *Environmental Assessment Act*, R.S.O. 1990, c. E.18, as amended. Consideration needs to be given as to whether any exemptions would be applicable to a particular project. Generally speaking, environmental assessments have to identify the potential environmental impacts of all aspects of the remediation project, as well as identifying proposed mitigation measures, the consequences of not performing the project, etc.

Various types of approvals may also be required for work in streams and lakes. For example, the removal of contaminated sediment from a stream using cofferdams and excavating equipment, may require approval under one or more federal or provincial pieces of legislation for example the *Navigable Waters Protection Act*, R.S. 1985, c. N-22, the *Environmental Assessment Act*, R.S.O. 1990, c. E.18 as amended, the *Public Lands Act*, R.S.O. 1990, c. P.43 as amended, and the *Conservation Authorities Act*, R.S.O. 1990, c. C.27 as amended.

Where the severity of the problem indicates that some type of remedial action will likely be necessary, consideration should be given to proceeding with applications for approvals and permits in the early stages of the project. Such applications can then proceed concurrently with assessment studies.

8.2 Federal legislation

A number of federal Acts apply to remediation work and may require permits and/or approvals in order to carry out remediation projects. Federal legislation can be divided into two groups: federal legislation applying to all proponents, and legislation and policies applying only to federal government departments.

8.2.1 Canadian Environmental Assessment Act

The *Canadian Environmental Assessment Act*, R.S.C. 1992, c. 37 (CEAA) is administered by the Canadian Environmental Assessment Agency. The Canadian Environmental Assessment Agency is an independent agency that reports directly to the federal Minister of the Environment. The CEAA requires that federal departments, including Environment Canada, agencies, and crown corporations conduct environmental assessments for proposed projects where the federal government is the proponent. It also requires environmental assessments when the project involves federal funding, permits or licenses.

The CEAA and its regulations set out the legislative responsibilities for the environmental assessment of projects that involve the federal government. The CEAA has four fundamental purposes:

- to ensure that the environmental effects of projects receive careful consideration before responsible authorities take action in connection with them;
- to encourage responsible authorities to take actions that promote sustainable development and thereby achieve or maintain a healthy environment and economy;
- to ensure that projects carried out in Canada or on federal lands do not cause significant adverse environmental effects outside the jurisdictions in which the projects are carried out; and
- to ensure that there is an opportunity for public participation in the environmental process.

An environmental assessment is required if a federal authority is required to exercise one or more of the following duties, powers or functions in relation to a project:

- proposes the project
- grants money to a project
- grants an interest in land to a project
- exercises a regulatory duty in relation to a project, such as issuing a permit or licence that is covered under the Law List regulation (SOR/94-636).

Similar to the Environmental Assessment and Review Process (EARP; the process used prior to the CEAA), the CEAA is based on the self-assessment of projects for environmental effects, by federal departments and agencies. The responsible authority may conduct an EA in the form of screening, class screening or comprehensive study.

Under a screening, a responsible authority has the greatest degree of management and flexibility over the scope and pace of the EA process. In cases where there is a sound knowledge of the environmental effects and appropriate mitigation measures for a group or class of projects, the responsible authority may be able to use all or part of a class screening report. The majority of projects covered by the CEAA will undergo an EA through a screening.

Under a comprehensive study, the responsible authority also retains a primary management role over the EA, but has more obligations than in a screening. These include the need to consider a wider range of factors, submit the comprehensive study report to the Agency for review, take public comments into account and consider the need for a follow-up program.

If the screening or comprehensive study identifies the need for further assessment, the project must move to a public review in the form of either a mediation or panel review.

8.2.2 Canadian Environmental Protection Act, 1999

A key aspect of the *Canadian Environmental Protection Act, 1999*, S.C. 1999, c.33 (CEPA) is the prevention and management of risks posed by toxic and other harmful substances. CEPA provides for the regulation of federal works, undertakings, and federal lands and waters, where existing legislation administered by the responsible federal department or agency does not provide for the making of regulations to protect the environment. In addition, there are provisions for the creation of guidelines and codes for environmentally sound practices and for setting objectives for desirable levels of environmental quality.

8.2.2 Migratory Birds Convention Act, 1994

The purpose of the *Migratory Birds Convention Act, 1994*, S.C. 1994, c. 22 (MBCA) is to protect and conserve migratory birds, as populations and individual birds, and their nests. The MBCA prohibits the disposal of any substances harmful to migratory birds in any waters or areas frequented by migratory birds (subsection 5.1(1)). In the context of identifying, assessing and managing contaminated sediments in Ontario, the MBCA may be applicable when dealing with the disposal of dredged material.

8.2.3 Fisheries Act

The *Fisheries Act*, R.S.C. 1985, c. F-14, regulates any activities that can potentially disrupt fish or fish habitat. The *Fisheries Act* applies to sediment remediation activities that may disrupt fish or fish habitat, which includes all manner of in-stream or in-lake activities associated with sediment remediation projects. Two sections of this Act are particularly relevant: Section 36 regulates the deposition of any substance (which would include contaminated sediment) which is deemed "deleterious" in waters frequented by fish. Section 35 regulates the alteration of fish habitat, including alteration, disruption or destruction of habitat (where habitat is defined as "spawning grounds and nursery, rearing, food supply and migration areas on which fish depend, directly or indirectly, in order to carry out their life processes").

8.2.4 Navigable Waters Protection Act (NWPA)

The *Navigable Waters Protection Act*, R.S.C. 1985, c. N-22 (NWPA) prohibits any work on, in, upon, under, through or across a navigable waterway. "Work" has been defined to include projects that involve the dumping of fill or the excavation of materials from the bed of navigable waters as well as dredging or disposal operations. An application for exemption is required for such projects. Prior to granting the exemption, Transport Canada reviews the implication of the project for potential impact on navigation.

8.2.5 Canada Water Act

The *Canada Water Act*, R.S.C. 1985, c. C-11, provides for management of the water resources of Canada, including research and the planning and implementation of programs relating to the conservation, development and utilization of water resources. Part II of the act specifically deals with water quality management and the pollution of waters.

8.3 Provincial legislation

Various Provincial Acts, administered by a number of ministries, may apply to sediment remediation activities.

8.3.1 Environmental Assessment Act

The *Environmental Assessment Act*, R.S.O. 1990, c. E.18 (EA Act) applies to projects being carried out by the Province, municipalities, or public bodies (for example, Conservation Authorities and the Ontario Realty Commission). These public sector actors may have obligations under the *Environmental Assessment Act* in respect of contaminated sediment management activities and should consult with the Ministry's Environmental Assessment and Approvals Branch.

8.3.2 Environmental Protection Act

The *Environmental Protection Act*, R.S.O. 1990, c. E.19 (EPA) regulates the discharge of contaminants and pollutants (including "spills") into the natural environment. The Act aims to protect and conserve the natural environment and to protect human health and plant and animal life from injury and damage and provides for the "repair" of any such damage. This Act has broad applicability to remediation activities.

8.3.3 Ontario Water Resources Act

The discharge of any material into water that impairs or may impair water quality is prohibited by the authority of the *Ontario Water Resources Act*, R.S.O. 1990, c. O.40 (OWRA). Subsection 1(3) of the Act sets out a list of what constitutes “deemed impairment”. This includes, but is not limited to, material or derivative which causes or may cause injury to or interference with any living organism that lives in or comes into contact with the water or soil or sediment that is in contact with the water; or which causes or may cause a degradation in the appearance, taste or odour of the water, or which causes or may cause injury to or interference with any living organism as a result of it using or consuming the water, soil or sediment that is in contact with the water or any organism that lives in or comes into contact with the water or soil or sediment that is in contact with the water.

8.3.4 Clean Water Act, 2006

The *Clean Water Act, 2006* S.O. 2006, c.22 takes a watershed-based approach to source water protection and addresses all sources of drinking water. Its purpose is to protect existing and future sources of drinking water. Source water protection is the first barrier in a multi-barrier approach to protecting the water in Ontario's lakes, rivers and underground aquifers, and it complements water treatment by reducing the risk that water gets contaminated in the first place. Where contaminated sediment impacts the quality of source water, the Act and associated regulations may require remediation actions to occur.

8.3.5 Brownfields Amendments (Record of Site Condition Regulation)

Amendments made to Ontario's EPA and OWRA, among other provincial statutes, in 2003 are commonly referred to as “the Brownfields Amendments”. Several documents are available that provide information on these amendments, including:

- Ontario Regulation 153/04 (as amended) Records of Site Condition – Part XV.1 of the *Environmental Protection Act*.
- Soil, Ground Water and Sediment Standards for Use Under Part XV.1 of the *Environmental Protection Act* (March 9, 2004)
- Procedures for the Use of Risk Assessment under Part XV.1 of the *Environmental Protection Act* (October, 2005).

This regulation and supporting documents have replaced *Guidelines for Use at Contaminated Sites in Ontario* (MOE, 1997). A key component of the Brownfields Amendments is the introduction of a Record of Site Condition (RSC), which is used to document the assessment and remedial work conducted at a contaminated property. For properties where sediments are a component of the risk assessment, the legislation and applicable regulations should be consulted for details.

8.3.6 Beds of Navigable Waters Act (R.S.O. 1990, c. B4)

The requirements of the *Beds of Navigable Waters Act*, R.S.O. 1990, c. B.4, may impact on projects involving beds of “navigable waters”. While the term “navigable waters” is not defined in this Act, it is a term which has been interpreted by the courts over the years. Title to the beds of navigable waters is restricted through grants by the Lieutenant-Governor. Ownership of lands bordering navigable waters does not provide right of use of the beds of those waters.

8.3.7 Public Lands Act

The management, sale and disposition of public lands, which includes the beds of most lakes and rivers as well as seasonally flooded areas, is controlled by the *Public Lands Act*, R.S.O. 1990, c. P.43. The Ontario Ministry of Natural Resources (OMNR) may define zones as open, deferred or closed for disposition. The *Public Lands Act* also regulates development, construction, or alteration of any public shorelands, which may apply to remediation projects. All shoreline construction work will require a Work Permit issued by OMNR under this legislation.

8.3.8 Conservation Authorities Act

The restricting or regulating of water through the construction of dams or diversions or depressions in rivers and streams and the placing and dumping of fill within the watershed is placed under the jurisdiction of the local Conservation Authority, under the *Conservation Authorities Act*, R.S.O. 1990, c. C.27. This Act would be broadly applicable to flow diversions such as coffer dams, channelling, etc. as part of the remediation project.

8.3.9 Lakes and Rivers Improvement Act

Approval for any work that consists of forwarding, holding back or diverting water (e.g., construction of coffer dams for stream remediation) is required from the OMNR under the *Lakes and Rivers Improvement Act*, R.S.O. 1990, c. L.3. Furthermore, the deposition of any substance or refuse into a lake or river or on the shore is prohibited by this Act.

8.3.10 Planning Act

The Provincial Wetlands Policy Statement, which was issued under the *Planning Act*, R.S.O. 1990, c. P13 addresses wetland protection and management within the land use planning process.

8.3.11 Mining Act

All aspects of mining activities within the province are regulated under the *Mining Act*, R.S.O. 1990, c. M14.

8.3.12 Nutrient Management Act, 2002

The purpose of this *Nutrient Management Act*, S.O. 2002, c. 4 (NMA) is to provide for the management of materials containing nutrients in ways that will enhance protection of the natural environment and provide a sustainable future for agricultural operations and rural development.

8.4 Municipal legislation and policies

Municipal legislation and policies will affect a project where shoreline or upland disposal is to be used. In these cases, municipal zoning or planning guidelines may have to be considered and taken into account. Since each municipality may have different requirements, the proponent is advised to contact the appropriate municipal office during the initial screening stage of the project. Contacting the municipal office will also permit the proponent to assess the need for public information sessions to facilitate public acceptance of the project.

8.5 Great Lakes Water Quality Agreement (GLWQA, 1978)

The Great Lakes Water Quality Agreement is an agreement between Canada and the United States to restore and enhance the water quality of the Great Lakes. The Agreement, first signed in 1972 and renewed in 1978, expresses the commitment of each country to restore and maintain the chemical, physical and biological integrity of the Great Lakes Basin Ecosystem and includes a number of objectives and guidelines to achieve these goals. In 1987, a Protocol was signed amending the 1978 Agreement. The amendments aimed to strengthen the programs, practices and technology described in the 1978 Agreement and to increase accountability for their implementation.

- Annex 2 of the Agreement relates to Remedial Action Plans and Lakewide Management Plans to control and remediate areas where "beneficial uses" have been impaired and specifies the need for source control programs to reduce loadings of Critical Pollutants.
- Annex 7 of the Agreement relates specifically to dredging activities and specifies that the two governments will develop and implement programs and measures to ensure that dredging activities will have a minimum adverse effect on the environment.

- Annex 12 relates to the presence of persistent toxic compounds and stipulates that the governments shall take all reasonable and practical measures to rehabilitate those areas of the Great Lakes adversely affected by these chemicals.
- Annex 14 of the agreement provides for the governments, in cooperation with State and Provincial Governments, to identify the nature and extent of sediment pollution in the Great Lakes System and subsequently develop and evaluate methods to remedy such pollution.

9 Risk management

Risk management is distinct from risk assessment; the latter is primarily scientific, the former includes risk assessment along with other non-scientific considerations such as societal and economic concerns. Good science alone does not yield good management, but is an essential prerequisite for good decision-making. For example, the “*range and significance of natural processes...must be adequately assessed prior to the selection, design and optimization of any management options for contaminated sediments*” (Apitz et al., 2002).

Application of the framework will assist with identification of actions that may be employed to improve environmental impairments caused by sediment contaminants. These improvements may include: no consumption advisories for public health or wildlife (i.e., guidelines and objectives not exceeded); healthy benthos, fish and wildlife populations (i.e., self-sustaining communities at the expected level of abundance when compared to reference conditions or, in the absence of community structure data, no significant water or sediment toxicity); normal rates of fish tumours, deformities and reproductive problems in fish, birds and mammals (i.e., rates not elevated above reference conditions); and, no restrictions on dredging activities (i.e., guidelines and objectives not exceeded).

This section of the document provides a brief overview of different processes and options for the management of contaminated sediments. This information has been reproduced from material originally provided in *An Integrated Approach to the Evaluation and Management of Contaminated Sediments* (MOE 1996). No new material has been added. As a result, this information may be dated and other sources that provide newer or more proven techniques may now be available.

9.1 Developing an action plan

9.1.1 Considerations governing sediment remediation

A sediment remediation plan is comprised of a series of carefully laid out steps designed to achieve a desired goal or objective. The most common goal is reducing sediment contamination to an acceptable level. The actions contemplated in the plan must be based on the ecosystem concept to ensure that short term gains do not cloud potential long term problems or that problems are not shifted from the aquatic medium to an upland site. In this regard, the consequences of each proposed action to be taken must be fully evaluated before the plan is adopted.

Since continued inputs of contaminants to an area to be cleaned up will be counterproductive to an effective remedial effort, the most essential feature of any cleanup plan is the control of contaminant sources to the area. All sources of contaminant input must be identified and, where possible, their contribution quantified in order to develop sound source control measures.

Following effective source controls, other sediment remedial actions may be taken to speed up recovery. There is evidence that natural restoration will usually proceed once the sources are controlled. This process, which relies on clean sediment covering over the contaminated material, may require several years. In areas where the supply of incoming sediment is low, other forms of in situ restoration may be needed to speed up the recovery process.

Within the last few years, various sediment clean-up technologies have undergone testing to determine the capabilities of removing and treating contaminated sediment. A summary of findings to date is provided in Section 9.2. These technologies have been tried on relatively small volumes of sediment which makes it difficult at this stage to assess their cost effectiveness on large projects.

Within Ontario, most sediment cleanup operations have traditionally involved some form of dredging and confined or upland disposal, and most of these have been in response to chemical spills. It is now clear that suitable new sites for the placement of large volumes of dredged material are generally rare and existing waste disposal facilities such as sanitary landfills are approaching or already at capacity in most areas.

Where suitable disposal sites are available, such as in the vicinity of major Great Lakes harbours, dredging and confined disposal may still be the preferred option for sediment cleanup, at least until better solutions are found. This is because much knowledge and experience has been gained using this technique and it is not restricted by the volume of material to be removed, as are some of the newer techniques which are now evolving towards a large production scale.

9.1.2 Setting a goal

The International Joint Commission has identified a number of "use impairments" in its "listing/delisting" criteria for Great Lakes Areas of Concern. These include:

- Fish tumours or other deformities.
- Bird or animal deformities or reproductive problems.
- Degradation of benthos.
- Restrictions on dredging activities.
- Eutrophication or undesirable algae.
- Restrictions on drinking water consumption or taste and odour problems.
- Beach closings.
- Degradation of aesthetics.
- Added costs to agriculture or industry.
- Degradation of phytoplankton and zooplankton populations.
- Loss of fish and wildlife habitat.

Sediments alone may not contribute directly to this extensive list of use impairments but, through the slow release of contaminants in some areas, may be a source of chemicals to the water column. To progress from a contaminated sediment problem to the restoration of designated uses in an area will require a strategy that involves a phased approach, likely over several years, to achieve significant improvements. It must be remembered, however, that a problem which has been in the making for decades may not be solved quickly. It is imperative, therefore, that any cleanup goal aimed at use restoration be based on a realistic schedule that incorporates source controls and the practical constraints of removing or covering over contaminated sediment until a desired concentration is achieved.

Another consideration of a practical nature is that current technology can only handle small volumes of material within reasonable costs. This suggests that it might be more practical and economically feasible to deal initially with the zones of high contamination, often referred to as "hot spots", while addressing the remaining portions of the area as financial and other constraining factors become more favourable. The important aspect of sediment management at this stage is to set realistic goals based on the practical nature of the options for achieving the goals, as well as the social, economic and environmental costs and benefits of achieving the goals. Such analysis must also consider the "do nothing option". The rest of this section describes some of the factors that may be considered in setting cleanup goals.

9.1.2.1 Factors to Consider in Setting Cleanup Goals:

The nature of the area and the problem

- The size of the area affected needs to be clearly defined since this will have a significant bearing on the remedial option chosen from both a cost and technology perspective.
- The uses the area is put to and the potential for this area to affect adjoining areas through the spread of contaminated sediments.

Uses may include protection of fisheries and benthic organisms. There is a need to consider both the toxic and bioaccumulative potential of contaminants. In previous sections, the need to look at a range of tests was indicated. This becomes critical at this stage since the severity of the effect will play a major role in arriving at the final decision.

From a human health perspective, compounds that are persistent and pose a threat to water supplies or fish and wildlife will be weighted differently from compounds that do not pose similar threats. In some cases recreational/aesthetic considerations may be the driving force in a cleanup study.

- The potential for recontamination must be examined from the point of view of existing and proposed land use and source controls. Existing and new industries

must incorporate features that will not lead to sediment contamination. It will be prudent to view this aspect not only from a local perspective but also from a broader watershed or regional perspective.

- There is a need to consider whether sediment removal will create additional problems, such as the exposure of historical contamination in deeper layers of the sediment. Care must be taken to ensure that the full depth of the problem has been adequately defined.
- The physical environment of the area needs to be considered. The potential for resuspension of contaminated sediment, with resultant contamination of adjacent or downstream areas will be an important factor in developing a remediation plan.

The nature of the solution

The solution to the identified problem in the area under investigation must be based on a clear understanding of the problem by all members of the decision making team. Science provides a large measure of the problem definition, but often cannot be comprehensive enough to provide answers to all the questions that may arise. The shortcomings are recognised, and methods such as risk analysis are relied upon to provide a structured basis for decision making. Members of the decision making team may experience difficulty in weighing the "apparent conflicts" that arise from considering all the results such as sediment chemistry, benthic surveys, toxicity testing, fish tissue residue, etc. This is especially troublesome when one assessment technique suggests a problem and the others do not. For this reason the framework provides a good basis on which to conduct an evaluation. The stepwise process of the framework eliminates the chances of arriving at erroneous conclusions.

In deciding on a rationale for cleanup, the cleanup objectives may not be realized over the short term and should realistically be viewed as longer term objectives, recognizing also that some of the uses identified may never have existed in the area.

Setting an outline for action

With the exception of spills, which must be cleaned up immediately, the most urgent need in environmental management is to protect the ecosystem from further abuse. Thus, source control must be the foundation of remedial action.

Consideration of remedial action in an area of contaminated sediment requires the development of a cleanup goal. This goal should be based on the "desired state of the environment" or developed in support of certain "attainable" uses. These need to be evaluated within the context of the whole watershed (i.e., a holistic approach). Where feasible, chemical guidelines provide a very convenient tool for setting cleanup goals, although these must be used with care since most chemical guidelines have been developed for broad use and may require some adjustment when applied to specific

sites and may, in some cases, be too conservative. Defining cleanup goals that are risk based (i.e. consider the pathways of the contaminants into the aquatic biota and its effect on the overall ecosystem health) differs depending upon whether the risk results are from direct or indirect exposure pathways. In setting cleanup goals to protect aquatic ecosystems, exposure pathways of the contaminants in the sediments should also be considered. Exposure pathways can be both direct, in that aquatic organisms are in direct contact with the sediments, or indirect, in that the organisms do not come into direct contact to the sediment, but are still exposed to the contaminants that originated from the sediment. The final goal could also include intermediate goals, since the achievement of the goal can be phased over time or over a sequence of activities.

The steps involved in developing a cleanup plan have been summarized below:

Development of Remediation Plan

- Base need for remedial action on biological effects and contaminant concentrations
 - severity of biological effects
 - ambient water and sediment quality
 - types of contaminants
- Determine effectiveness of source controls for all sources
 - ability to control sources will affect final remediation target
- Need to consider local land use and local “best use”
 - goal should be reasonably achievable
 - compatible with existing land uses
- Consultation with stakeholders/public
 - stakeholder involvement to ensure “best achievable goals” are being considered
 - consideration of existing goals (e.g. RAP/PAC, publicly established)

The ideal cleanup goal will always be the level that provides for the protection of all sediment uses. In most cases the target will be determined by the local background, ambient values, or based on the results of an Ecological Risk Assessment, since these are the practical limits to cleanup. However, cleaning up to this level will not always be feasible, especially when the area under consideration is large or where there are ongoing sources of contamination. Such areas may require a multi-phased approach, with a lengthy time frame, to achieve source control before any remediation work is undertaken.

In some areas, the ideal cleanup opportunities present themselves and these should not be overlooked. For example, Collingwood Harbour remedial efforts (i.e., dredging) were

assisted by the presence, on-site, of facilities designed for navigational dredging that were ready to accept the sediment. Such ideal situations may not exist in all areas; however, all options must be explored.

If cleanup cannot be accomplished in a single operation, then a sequence of operations should be planned and for each one a plan should be developed on what is to be achieved within a given time frame. The plan should be based on the following:

- The cleanup plan must be based on realistic goals. If the goal is too low or cannot be achieved within practical economic limits, then the effort will be of little practical value. The cleanup goal must be compatible with the prevailing land and water uses in the area. The existing uses of the area will influence the final remediation target. For example, an area receiving stormwater runoff from an urban area will require a different cleanup target from one influenced mainly by rural activities.
- The plan should be compatible with proposed local land uses. The proposed uses of the adjoining areas and perhaps the local watershed will influence the remedial plan. For example, an area that is used exclusively for recreation will be considered differently from one with mixed uses.
- The plan must consider the quality of sediment entering the area from remote sources such as upstream areas. This is especially important in enclosed areas such as harbours where most of the sediment that enters the area is deposited. It will be counter-productive in such cases to cleanup existing sediments when the problem could recur over a short time.
- The nature of the contaminants will play a role in target setting. The significance of potential health and environmental effects of contaminants in sediments is determined to a large extent by the types of contaminants encountered. Some compounds, such as the persistent organics, will pose a greater environmental threat and will therefore elicit a different response from decision makers compared to certain metals.
- The goal will also be influenced by cleanup technology, suitable disposal sites for sediment if the removal option is selected and equally importantly, appropriate funding.
- A phased approach would require setting interim targets that can be achieved over a prolonged time frame. Such an approach should clearly identify a number of milestones to ensure that progress is being made.

None of the remedial options are free of risks and any contemplated action will have certain benefits as well as certain negative impacts. Properly designed studies will highlight potential problems and provide some indication of the magnitude and

significance of potential impacts. In effect, such studies will define the current state of the sediment environment.

In many areas, contaminated sediments have been an historical problem which needs to be resolved. The resolution starts with turning off the sources and, with the help of natural processes, improving the situation. In some cases the problem is so severe that additional efforts will be required to speed up the environmental healing process. Those involved in remediation should be aware that restoration to "pristine" conditions is usually an untenable and often unattainable goal, as is the expectation that immediate results can be obtained. As noted earlier, many areas will require an extended period of time, even with active remedial efforts.

On the other hand, a "do nothing" approach just for the sake of ignoring a problem is not acceptable. In the past, decision making was often delayed as additional information was sought to answer questions as they arose. In many cases, there are no immediate answers to such questions and if we try to answer every scientific question that arises, decisions will seldom be made. Therefore, it is imperative we work with the types of information gathered, recognizing that definitive answers are rarely provided given the current state of the science in information gathering. The framework has been designed such that data gaps can be identified as part of the initial decision-making process, thus avoiding timely and costly delays.

Efforts should be directed towards addressing the essential questions: How much cleanup is needed? Where is it needed? How much will it cost? How can it best be done? The framework outlined in this document addresses the first two questions, and the section that follows addresses remedial options.

9.2 Sediment remediation options

9.2.1 Remediation options

In most cases the choice of remedial options will depend on the nature of the contaminants and the severity of contamination. Where immediate cleanup is required as a result of high concentrations and severe biological effects, the range of options may be more limited than in areas where contamination is less severe.

Nevertheless, a broad range of options is available for dealing with contaminated sediments and these have been summarized in Figure 7. These range from simple removal technologies to elaborate in-situ treatment. Predictably, as the complexity of the treatment increases, so do the associated costs, and the most elaborate methods are usually also the most expensive.

Sediment remediation technologies fall into three broad categories: A) natural remediation; B) removal, sometimes followed by some type of treatment and; C) in-situ

treatment. Currently, only the first two options have been used on a large scale and can be considered as proven technologies.

The selection of a remedial option requires careful consideration to ensure that any actions taken do not exacerbate the problem. In some situations, in-situ contamination may pose only a minor threat to organisms when in-place, particularly if the material is isolated from the water column, but may become a major concern when disturbed. Removal operations may resuspend contaminated material, thus potentially increasing its availability to aquatic organisms. Removal may also expose deeper layers of contaminated material. Finally, removal will present other concerns since in most cases this will require some type of secure disposal or additional treatment.

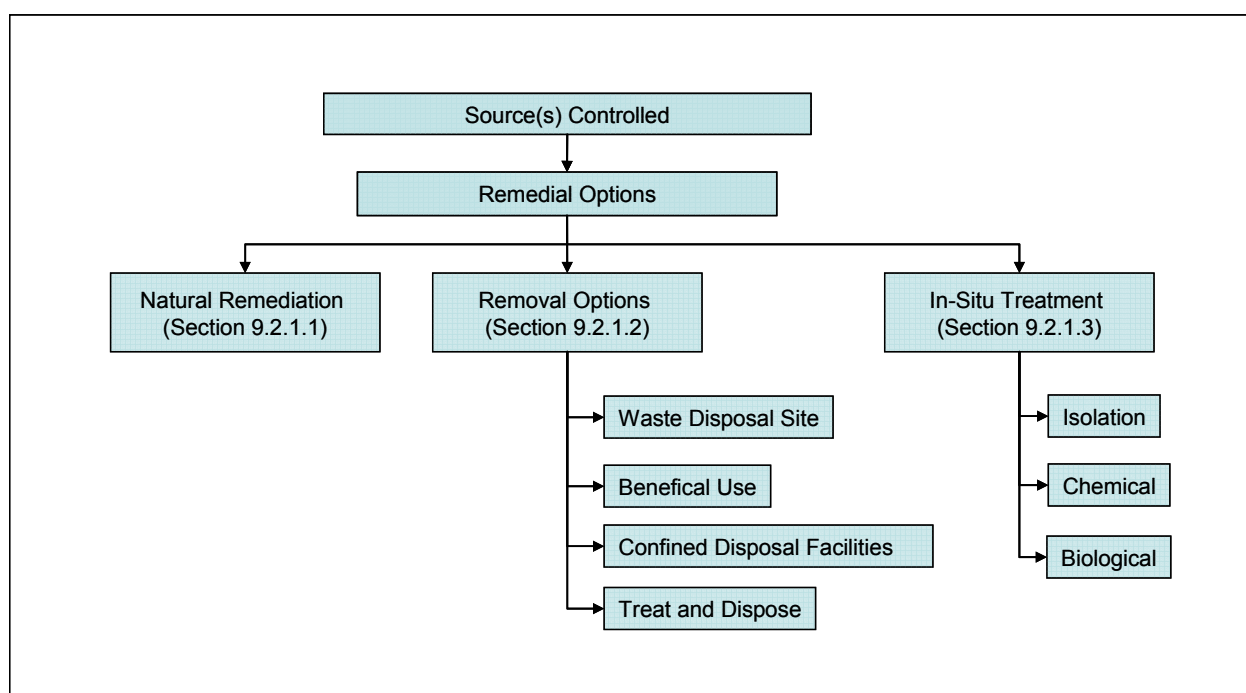


Figure 7: Sediment Remediation Options

9.2.1.1 *Natural remediation (monitored natural recovery)*

Natural remediation can be considered in those situations where the problem is not so severe that material must be removed immediately. Natural remediation is a preferred option where large areas of relatively low contamination are considered. The economic costs of removing and treating such areas usually make this the most viable option. However, in all cases, this option assumes that effective source control has been achieved.

Natural remediation consists of leaving the sediments in-place, to be buried by newer, cleaner material. "Treatment" relies on the finding that many of the contaminants in sediments undergo changes in the sediment. Metals, for example, will often mineralize

over time, creating insoluble compounds. Organic compounds will usually degrade over sufficiently long time periods.

In some cases, leaving the material in place may effectively reduce the availability of contaminants. In particular, metal availability can be reduced through natural processes such as diagenesis and mineralization. Most organic compounds, even persistent organics, will decompose as a result of microbial action, though in some cases this process may be exceedingly slow. If the contaminated material is isolated from the water column by a cleaner surface layer, then it would pose little threat to aquatic organisms.

Therefore, for natural remediation to be an effective option the sedimentation rate must be high enough that the material will be covered to a depth required to isolate the material from the water column in a reasonable period of time.

Though often equated with the "do nothing" option there are significant differences between the two. The "do nothing" option does not consider natural burial, rates of sedimentation or severity of contamination. It is most often used where it is not technically feasible or practical to undertake remediation. Natural remediation is not a "do nothing" option but requires evaluation of existing conditions to ensure that sediment burial is achieved within a predefined period of time. Natural recovery may require ongoing care and oversight to ensure environmental conditions are not worsening and recovery is taking place. The application of administrative conditions or rules may be required to ensure that recovery is not negatively affected by human activities.

It is important to determine the dynamics (transport) of both contaminated sediments and clean sediments in order to determine the acceptability of this option. If clean sediments are being continually deposited over historically contaminated sediments and point source controls have been achieved, this may be the preferred option. If however, contaminated sediments are being slowly or episodically removed and re-deposited elsewhere, other remedial options may have to be considered. The advantages and limitations of natural remediation have been summarized below:

Remediation Options

Natural Remediation

Advantages:

- least likely to resuspend contaminants
- no sediment loss due to handling
- no disposal problems
- natural mineralization of metals
- natural decomposition of organics

Suitable for sediment with:

- chronic biological effects
- low bioavailability of sediment contaminants
- low potential for biomagnification

Not suitable for sediment with demonstrated:

- acute toxic effects
- high sediment contaminant bioavailability
- strongly bioaccumulative compounds

Requirements:

- depositional area
- sufficiently high deposition rate
- low potential for disturbance

9.2.1.2 Removal and disposal or treatment

All remediation activities that involve removal of sediments require some type of dredging operation. Dredging can be accomplished using either mechanical or hydraulic dredges. Mechanical dredges include the familiar clamshell dredge, as well as a variety of other modified buckets and may be deployed from the shore or barge-mounted. All have the advantage of being able to remove sediment at in-situ densities with little additional entrainment of water. However, these dredges typically result in high turbidity from re-entrainment of settled material. For remediation of contaminated sites, this is often highly undesirable and necessitates the use of additional measures to contain the suspended material, such as silt curtains. In many cases, physical conditions within the water body preclude the effective use of these mitigation measures and alternative methods of dredging, such as hydraulic dredges, may be required.

Hydraulic dredges typically operate with a much lower turbidity than mechanical dredges. Hydraulic dredges are usually comprised of some type of cutting head that loosens the sediment and mixes it with water, and a pumping system that pumps the slurry either to a holding tank such as a barge, or through a system of pumps to a shore-based holding area. Hydraulic dredges typically entrain large volumes of water in order to achieve a pumpable slurry. Where clean sediments are being removed, as in

the case of navigational dredging, the hopper or barge is allowed to overflow the excess water. For remediation of contaminated sediments, use of hydraulic dredges will necessitate treatment of entrained water since, due to the presence of contaminants, the water cannot be overflowed.

The removal equipment can vary in size, depending on the needs or access restrictions of the area. Most removal equipment has been designed to operate at practical depths of up to 10 m. In deeper water, removal of contaminated sediment may be limited by the availability of suitable equipment. While clamshell-type dredges are theoretically capable of operating at any depth, their accuracy can be substantially diminished at greater depths.

Considerations/limitations

Loss of sediment during the dredging operation is a major concern with removal of contaminated sediment. Loss of material to the water column has the potential to distribute contaminated fine material over a broad area, which can also heighten the bioavailability of contaminants to aquatic organisms. The major concern is to control such losses, either through the use of special cutting and dredging devices that minimize the amount of material that is resuspended, or through the use of additional equipment such as silt curtains that can limit the spread of resuspended material. Both methods have their limitations. Special cutting heads for use with suction hopper dredges will require additional facilities for treatment of entrained water, since these types of dredges typically require high water content in order to remove the material. Silt curtains are suitable for use only in areas of little or no current and during calm weather.

In flowing water situations removal operations may be carried out in the "wet" or "dry". The type of removal operation will depend on the size and flow of the stream or river. In large rivers sediment removal will be carried out using the same equipment as used in standing water (dredges) though containment devices such as silt curtains are often unsuitable in these situations. In smaller rivers and streams, sediment control measures may require the placement of temporary barriers to control resuspension. In some cases the isolation of the contaminated area through the construction of cofferdams and dewatering of the site may be the most preferable.

The actual removal of contaminated sediment can result in a number of adverse effects in shoreline areas through losses of material. Such problems are heightened each time that the material is handled. Such activities are also disruptive to shoreline and navigational uses of the area as heavy equipment is brought in. Most areas will require a shoreline staging area, as well as temporary materials storage and sediment dewatering facilities. Some of the newer technologies will require extensive areas set aside for considerable periods of time for treatment facilities, and most will require transport to a final disposal site. Each of these steps has the potential to result in adverse effects from loss of material. Proper containment devices will be required as well as protective measures for site workers.

Contaminated sediments present special transportation problems. Materials dewatered and treated on the site should be transported in covered containers to minimize losses as wind-blown dust. However, wet materials moved off the site for treatment or disposal must be transported in sealed trucks or other containers.

Temporary or final storage needs must be determined on a site specific basis since this depends on the material involved (i.e. how toxic), the volume, treatment needs, and any possible end-uses after treatment. Storage facilities should be designed to address these specific concerns.

A number of new designs for removal equipment that minimize sediment loss have also been tested, including such specialized devices as the pneuma-pump. The following list of treatment options considers the technologies only in terms of broad categories. Within each category a number of different technologies exist, many of which are proprietary. Sources for detailed information on specific processes are available in a number of publications, including:

- *Review of Removal, Containment and Treatment Technologies for Remediation of Contaminated Sediment in the Great Lakes.* (Avrett et al. 1990).
- *Sediment Treatment Technologies Database. 2nd Edition.* (Wastewater Technology Centre 1993)
- *Screening Guide for Contaminated Sediment Treatment Technologies.* Environment Canada, St. Lawrence Centre (EC 1993)

Once the contaminated sediment has been removed, a number of disposal options can be considered. Depending on availability and local acceptability, the material could be disposed of in confined disposal facilities (CDFs), in landfills, or hazardous waste landfills. In many cases, however, to keep disposal costs down, pretreatment of the material (i.e. prior to disposal) will be required. The advantages and limitations of this option are summarized in the inset below:

Remediation Options

In-situ Treatment

Advantages:

- eliminates need to remove and treat/adequately dispose of toxic sediment
- reduces sediment losses through removal and handling

Suitable for sediments with:

- contaminants for which treatment technologies exist
- suitable physical conditions (e.g., for capping)
- ongoing existing sources
- where removal is impractical

Not suitable for sediments of:

- unstable bottom conditions or untreatable chemical compounds
- where rapid removal/isolation required

Requirements:

- contaminants must be amenable to chemical treatment
- must effectively complex/isolate contaminants within a suitable time frame

Dredging, Dewatering, Solidification and Disposal

This option involves the removal of the sediment, subsequent dewatering and solidification and disposal in an acceptable site.

- primarily for dredging of sediment defined as hazardous material.
- removal usually requires hydraulic dredging or, if conventional dredging is used, additional containment devices such as silt curtains will be necessary.
- hoppers cannot be overflowed and entrained water must be treated prior to return.
- mechanical dewatering consists of: filters, centrifugation, or thickening.
- evaporation dewatering: requires energy input for evaporation and will require treatment of off-gases.
- choice of dewatering depends on amount of sediment, available temporary storage for material awaiting processing, and desired consistency of the material (depends on disposal alternative).

Removal and Treatment

Most of the post-removal treatment technologies currently developed or being tested fall into one of three major groups: 1) Extraction processes, which involve dissolution of the contaminant in a recoverable fluid; 2) Immobilization processes which chemically fix or alter the contaminants and; 3) Thermal processes which use heat to break down the chemical bonds of the contaminants or to vitrify contaminants and sediments into a solid mass. The advantages and limitations of this method are summarized below:

<u>Remediation Options</u> <u>Removal and Treatment</u>
<p>Advantages:</p> <ul style="list-style-type: none">- removes toxic sediments- removes/ destroys toxic compounds- eliminates need for expensive containment (still requires on-land disposal) <p>Suitable for sediments with:</p> <ul style="list-style-type: none">- acutely toxic effects- high sediment bioavailability- highly biomagnifying compounds- high sediment resuspension- compounds for which removal/destruction technologies exist <p>Not suitable for sediments of:</p> <ul style="list-style-type: none">- suitable for most areas though too expensive to consider for areas of low sediment toxicity or bioavailability <p>Requirements:</p> <ul style="list-style-type: none">- means of minimizing turbidity and loss during removal- large areas set aside for extended time for equipment and stockpiling of sediment

Few of the methods proposed or tested attain total destruction or removal. The most efficient technologies currently range from 80% efficiency for metals, to in some cases 99% efficiency for organics, though typically these are much less. However, in most cases, the residue will still contain contaminants and only in those cases where concentrations are at or below MOE fill quality guidelines for Lakefilling (MOE, 2003) can the material be safely returned to the aquatic environment. In most cases, the

material will have to be disposed of upland in accordance with MOE requirements, such as the *Environmental Protection Act*, O.Reg. 461/05 .

Dredging and Washing/Extraction

Following dredging, the materials are subjected to washing with various solvents that will preferentially bind target contaminants. The solvents are subsequently separated from the sediment. In theory, the cleaned material can be returned to the site, though in most cases it will have to be disposed of at an appropriate upland site. However, since the material will have lower contaminant concentrations at the end of treatment, a larger number of disposal options will be available.

- contaminated sediment is washed in a suitable solvent to remove the contaminants. The solvent is later recovered.
- solvents can be selected to remove either organics or inorganics (because of the different solvents required for each, it does not appear to be feasible to do both at the same time and removal of both will require multiple steps).
- requires a storage area and a treatment facility (i.e., tank of some kind).
- most processes require multiple extraction cycles.
- requires further separation of contaminants from solvent and final disposal of contaminant concentrate.

Dredging and Incineration/Thermal Destruction

- The material is dredged, usually dewatered to some degree and incinerated in high temperature incinerators.
- are among the more effective options for destroying organic contaminants. (up to 99% efficiency of removal)
- incinerators include: fluidized bed, circulating bed combustor, high-temperature slagging, infrared, multiple hearth, plasma arc, Pyretron, and rotary kiln.
- not effective for metals - volatile metals like mercury and lead may require additional steps to ensure removal from flue gases. Incineration can also change oxidation states of some metals, making metals in the final product more mobile.

Dredging and Vitrification

The material is dredged, dried to some degree and treated. The treatment uses high-voltage graphite electrodes to melt material. The molten material then cools to a solid glass-like material.

- does not measurably leach organic or inorganic contaminants.
- energy costs are high (depending on water content) as are operating costs (consumable electrodes).
- may require flue gas collection and treatment since process volatilizes semi-volatile and volatile organics.
- bench-scale tests confirmed better than 99% efficiency in PCB destruction.

Dredging and Low-temperature Thermal Stripping

Sediments are heated to relatively low temperature (ca. 350°C) to remove volatiles and semi-volatiles. The volatiles are condensed and the liquid cleaned and filtered through activated carbon.

- the dry, dust-like sediment will contain materials not driven off by low temperature.

Dredging and Reductive Dechlorination

Sediments are heated in the presence of a reducing agent such as hydrogen to dechlorinate organic contaminants.

- suitable for only a certain number of organic compounds (currently tested only for PAH and PCBs).
- better than 99% efficiency for those compounds tested (PCB and PAH).
- has currently been tested only in pilot and bench scale tests.

Dredging and Biological Treatment

Uses biological methods such as slurry-phase biodegradation. Most available methods rely on bacteria to decompose organic contaminants, either under aerobic or anaerobic conditions. These methodologies may require inoculation /addition of bacteria.

Dredging and Chemical Fixation (Immobilization, Solidification/Stabilization)

These involve a number of methods to limit contaminant mobility through introduction of a chemical fixative. While most require removal of the sediment, some procedures have been proposed for in-situ fixation.

- most effective for treating heavy metal contamination.
- involves injecting a solidifying agent (e.g., cement or lime) together with an additive to prevent organics from interfering with the solidification process.

Dredging and Chemical Treatment

Involves treatment with any of a number of chemicals to neutralize, fix, or alter contaminants in sediment.

- chelation uses chelating molecules to bind and restrain metal ions from forming ionic salts.
- efficiency is variable, depending on chelating agent and dosage.
- other processes include oxidation of inorganics, nucleophilic substitution.

Dredging and Multi-phase treatment

This category of treatment comprises a combination of the above treatments.

9.2.1.3 In-situ treatment

A limited number of options are available for in-situ treatment of contaminated sediment (the advantages and limitations of this option are summarized below). In-situ fixation methods work mainly through aiding natural remediation processes such as decomposition of organics in order to reduce, though not necessarily eliminate, sediment contamination.

Capping

The procedure involves covering existing contaminated sediment with clean material.

- unless capping material is less dense than material being capped, the materials may sink through.
- potential for erosion of cap materials
- reduces navigable depth and precludes future dredging.

In-Situ Fixation/Stabilization

Involves the injection of chemicals/additives that will either bind with contaminants to effectively remove them from circulation, or that enhances their decomposition.

- this is at present only in the developmental stages and has not been demonstrated in full scale.

Capping-Lakefilling (Overfilling)

This is similar to capping except that the area is isolated and the cap extends to the surface to create a lakefill.

9.2.2 Selection of remedial option

Once the available options have been identified the next task is to determine which are most suitable for the site in question. Initially, it needs to be determined whether active remediation is a realistic or feasible goal. Some areas, for example, may be prohibitively expensive and may best be left to natural remediation.

The evaluation of remedial options should include:

- level of contamination and severity of biological effects (some options are not suitable for heavily contaminated/ acutely toxic sites)
- volume and type of material to be remediated.
- physical factors such as navigational use.
- suitability of treatment(s) to the type(s) of contaminant(s) (where different classes of contaminants are involved, e.g., organics and metals, treatment may involve different processes for each that may have to be done in series).
- effectiveness of remediation and /or treatment (i.e., will the method remove only some or all of the contaminant of concern; will additional treatment or some type of confined disposal be required; will some steps have to be repeated).
- costs for remediation, including removal, treatment and storage.
- mitigation procedures required.
- potential for reuse of material.
- potential disruptions to current uses of the area (e.g., will navigational routes be affected, disruption/ destruction of fish habitat).
- length of time required for the project.
- public acceptance of the option.

A table should be prepared that lists the various options and the evaluation criteria. The most suitable option would then be identified through determination of which option not only meets the remediation target, but also most closely satisfies the priorities developed for the particular area.

9.2.3 Evaluating Cleanup Options

In evaluating the different options for sediment cleanup, consideration must be given to the "non invasive" option of natural remediation which, as mentioned earlier, relies on clean sediments depositing over the contaminated area through natural processes. The natural remediation option may be the preferred option in those cases where other forms of remedial action may result in a worsening of the situation by making contaminants more readily available through resuspension and dispersion. The natural remediation option would best be considered in depositional areas such as harbours where incoming clean sediments will form a cap over the contaminated material. This will apply to those areas where the sources have been controlled.

A convenient way of summarizing information to be used in deciding on a remedial option for contaminated sediment could be through construction of a matrix as shown in the following hypothetical case.

Evaluation Parameter	Identified Remedial Options				
	A	B	C	D	E
Cost Effectiveness	H	M	L	M	H
Uses Ecosystem Principles	M	L	M	L	L
Social Acceptability	M	H	M	L	L
Technical Flexibility*	L	M	H	H	H

*(Based on type and quality of material)

H = high ranking, M = medium ranking, L = low ranking

Prioritizing Options:

These have to be developed on a site specific basis, since local priorities will differ. However, cost and technical feasibility are typically major determining factors, and further evaluation of an option is usually not warranted if the option is technically unsuitable or the cost is prohibitive.

Cost Effectiveness:

Cost effectiveness refers to the financial costs of achieving the desired or stated objectives on sediment cleanup for each of the options being considered.

Costs must include all facets associated with the option, including: equipment, mobilization, removal, treatment, residuals disposal, etc. This has to be done for each phase of a multi-phased operation and must also include any post cleanup costs associated with management of material removed.

Ecosystem Principles:

The basic element in such principles is that extreme caution is used to ensure that a sediment cleanup operation does not result in the transfer of contaminants to another area where they pose a threat. In most situations, when sediment is removed from water it is placed in a confined disposal facility along the shoreline or is disposed of at an upland site.

Social Acceptability:

This includes public response to the measures being proposed as reflected through public advocates on the decision making team. Concerns may be related to the cost of the project, the effort and anticipated accomplishments. For example, Is it a partial or full cleanup effort? Will it take an inordinately long time to effect? Will the proposed

solution create problems in other areas (e.g. will material have to be disposed of on land with potential to affect existing land use or environmental quality?).

Technical Feasibility/Implementability:

The method selected must be capable of dealing with the problem without generating any problems of its own, e.g. if dredging is required, then the type of dredge selected should minimize the loss of material through resuspension and dispersion. The size and type of equipment used must fit the problem and the area.

9.3 Remediation plan

The final step, once the remedial options have been identified, is the development of a remediation plan. This will identify the impaired uses, the remediation targets, and the means of achieving these targets. Included in the latter should be any mitigative measures needed to ensure that any adverse effects of the remediation are minimized. This step will also need to identify the means of disposal of the material, and the final best use for the area.

The final remediation plan is based on the results of both the sediment and biological studies and is developed in conjunction with socio-economic considerations. The steps to development of a remediation plan are listed below:

- 1) Identify both biological/chemical impacts and socio-economic impacts (such as impaired uses, etc.)
- 2) Determine area to be cleaned up. This is based on both scientific and socio-economic criteria.
- 3) Determine options for remediation. This should list all suitable options.
- 4) Identify the benefits and costs of each option on both an environmental and an economic level.
- 5) Determine the most appropriate cleanup strategy, which may not always be the most desirable from a purely scientific perspective.
- 6) Develop a detailed plan for remediation, identifying all the major steps and a timetable for implementation. These include the details of removal or in-situ treatment such as schedules, areas, volumes to be treated or removed, temporary and permanent disposal sites, etc.

It is at this level that the truly hard choices must be made. At the earlier steps in this procedure, the scientific criteria have been determined, and the best environmental solutions have been developed. It is unfortunately true however, that the best scientific solutions are not always the most practical. The costs of each option must be weighed against the benefits, and the choice made may not be the best from an environmental perspective. Although it is beyond the scope of this document to describe the socio-economic process associated with sediment cleanup, it is nevertheless a major

component of the decision making process and the proper expertise must be obtained to conduct such an assessment.

A number of remediation options are suitable only for certain types of contaminants, or are practical only for low volumes of material. If removal and off-site confinement are considered, then it is necessary to evaluate the safety and integrity of the confinement site. Not all material is suitable for such storage. Dredged material, for example, should be evaluated according to Fill Quality Guidelines for Lakefilling in Ontario (MOE, 2003) to determine suitability for disposal in sites other than registered landfills or hazardous waste sites. This will reduce disposal costs as well as conserve scarce disposal areas.

9.4 Sediment cleanup

9.4.1 Implementation of remediation plan

Once the remediation plan has been developed and approved by all concerned, the plan needs to be implemented. During actual implementation of the plan, all efforts need to be directed towards ensuring that the approved plan is followed, with as little deviation as possible. Some deviations will always be necessary, as unforeseen situations arise. A mechanism for resolving such problems should also be in place, to ensure that the task can be completed quickly. In many cases, the danger of contaminant release or escape (with often broad dispersal) is heightened by prolonging the construction /remediation period.

In all cases, once remediation is underway, a site manager should be present at the site to ensure the remediation plan is being followed, and to deal with unforeseen situations as these arise. The site manager should have overall responsibility for the cleanup actions.

In many cases the specific precautions taken to minimize adverse effects will depend on site-specific considerations. This will be dictated by considerations such as the actual method of removal, the methods of treatment and the means of disposal.

Examples include:

- special handling procedures
- special dredging techniques
- treatment of overflow water from hopper
- silt curtains or other sediment containment devices

Of the intrusive remediation options, in-situ remediation is usually the least disruptive. Depending on the methods used, some disturbance of the surface layers can be expected, though the effects will usually be localized to the immediate area.

During cleanup, a site supervisor should be present at all times to monitor activities at the site. The supervisor needs to be fully aware of the potential problems and fully appraised of the details of site cleanup.

9.5 Post-remediation

9.5.1 Monitoring effectiveness of remediation

An essential element of any cleanup operation is a measure of its effectiveness. Assessment of the effectiveness of the operation should include all of the parameters/uses that were identified as impaired, as a measure of achieving the remediation target. The study can also include additional tests, which may indicate whether the cleanup has resulted in additional effects that may not have been anticipated.

Where chemical criteria were used as targets, biological monitoring should be employed as well. For example, monitoring may reveal that there are no biological impacts despite sediment concentrations in excess of the LEL.

The assessment should also determine whether the "local best use" identified earlier is now achievable. This may not be readily apparent, since any area that has undergone remedial action will require a period of time to stabilize.

Finally a decision must be made on the long term monitoring of the area: how frequently, and for how long such monitoring should be continued, and when can it be stopped, are questions that need to be addressed. In many cases, such long term monitoring should be instituted some time after the initial post-cleanup assessment in order to measure effects after the area has stabilized. A time period of three to five years between sampling is recommended.

10 Concluding remarks

The purpose of this document is to provide guidance for identifying, assessing and managing contaminated sediments in Ontario. The document provides step-by-step science-based guidance for assessing risks posed by contaminated sediment. The document is intended to be sufficiently prescriptive to standardize the decision-making process, but without using a "cook book" assessment approach that would fail to acknowledge the influence of site-specific conditions on the outcome, nor allow for appropriate use of best professional judgement. Assessing and managing contaminated sediment is complex and is best approached using a multi-disciplined team of qualified persons.

For further support and guidance on the components of this document and on the Provincial Sediment Quality Guidelines outlined herein, please contact the Ministry of Environment's Standards Development Branch. For queries pertaining to the monitoring and assessment of sediments, contact the Ministry's Environmental Monitoring and Reporting Branch, and for site specific sediment issues contact your closest regional Ministry of Environment office.

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APPENDIX A:
Protocol and Rationale for setting Provincial Sediment Quality Guidelines

Protocol for setting Sediment Quality Guidelines

Rationale for setting Sediment Quality Guidelines

In developing guidelines to provide adequate protection for biological resources, the Ministry has attempted to ensure that the methods employed consider the full range of natural processes governing the fate and distribution of contaminants in the natural environment. Since benthic organisms respond to a variety of stress-inducing factors they are, in essence, integrators of all the physical, chemical and biological phenomena being experienced in their environment and these organisms should form the basis of any method used in setting sediment guidelines.

Because individual species may respond differently to stress-inducing factors it is very difficult to study a specific organism (e.g., a sensitive species) with the hope of developing guidelines that will protect the rest of the community. Sensitivity to chemical contaminants has not been fully evaluated for different benthic organisms and most sediment bioassay work has been concerned mainly with a few selected species (e.g., the mayfly *Hexagenia*). While the mayfly has traditionally been used as a "sensitive" indicator organism for factors such as low dissolved oxygen, its sensitivity relative to other benthic organisms has not been clearly established for chemical contaminants. Therefore, in developing PSQGs, the Ministry has not relied on single-species data.

Similarly, a method that relies heavily on those species that are known to be extremely tolerant of contaminants in sediment cannot result in guidelines that will adequately protect less tolerant members of the aquatic community. It has been demonstrated that some populations can adapt to varying levels of environmental contamination with increasing tolerance to these contaminants occurring in succeeding generations. This can present difficulty in laboratory studies of reared populations since these may lack the genetic diversity found in natural populations and responses may not be consistent with those observable under field conditions. There is also concern over placing heavy reliance on laboratory data as in most situations contaminants in sediments exist as mixtures of various substances. Laboratory tests generally tend to examine the effects of single substances, and laboratory data can be difficult to apply to field situations.

In developing the protocol for setting sediment quality guidelines, the Ministry considered a number of different approaches developed by state and federal agencies in North America that employed various degrees of biological assessment. The various suggestions for the development of sediment quality guidelines can be summarized in five approaches as possible means of setting sediment quality guidelines. At present, no single approach can adequately account for all the factors that operate in natural sediments and each of the five approaches has positive attributes as well as limitations with regard to the development of biologically based guidelines. The rationale used in setting sediment quality guidelines includes a number of considerations which are detailed below. These considerations provided the basis for selecting the best method or combination of methods for the development of Provincial Sediment Quality Guidelines (PSQGs).

1. Sediment quality guidelines should consider a range of contaminant concentrations that is wide enough to determine the level at which ecotoxicological effects become noticeable. This can be achieved most effectively by looking at a large number of organisms under the widest possible range of contaminant exposure. Only then can

the appropriate ecotoxicological level be adequately determined. A restricted range may result in the setting of guidelines that are not reflective of actual ecotoxicological effects on organisms and as such may be overprotective. This is especially important where the range of effects used may not cover the entire tolerance range of the species in question.

2. PSQGs should be based on cause-effect relationships between a specific contaminant and benthic organisms since it is necessary to demonstrate that at a certain concentration a contaminant results in adverse effects on benthic organisms.
3. PSQGs should account for contaminant effects in a multi-contaminant medium. Since contaminated sediments usually consist of mixtures of substances, the presence of a number of different contaminants, any or all of which may affect the response of the organisms to the contaminant being investigated must be considered. Since exposure to combinations of contaminants may result in different responses than exposure to a single contaminant (through either synergistic or antagonistic effects), these effects must also be considered. A PSQG method must incorporate this feature into the derivation of a number for specific contaminants.
4. PSQGs should consider chronic effects of contaminants on aquatic biota since these can affect the long term viability of aquatic organism populations. Methods that consider only acute effects do not offer adequate protection, since sediment concentrations reflect long-term conditions and are not subject to the extreme temporal variability of water column contaminant concentrations.
5. The PSQGs should be capable of incorporating and accounting for the range of environmental factors that could have a bearing on the presence or absence of organisms in a given area. Contaminant behaviour and the organisms' well-being are governed by a variety of natural physical, chemical and biological processes. If these processes are not accounted for in a PSQG method then the resulting guidelines will be unrealistic. For example, organisms may be absent from a given area not because of the level of contaminants but because of unsuitable habitat, low dissolved oxygen, or interspecific competition. In formulating a guideline it is essential that these factors be considered along with the chemical data. If they are not considered, the numerical value obtained would not necessarily be protective of aquatic species. This will also reduce the need for site-specific guidelines, since a full range of environmental conditions will have been covered.

Approaches to developing Sediment Quality Guidelines

As part of the sediment guideline development process, the Ministry has carried out an extensive literature review of possible approaches to the development of sediment guidelines. This effort has resulted in the selection of five potential approaches for this purpose. These are:

1. Sediment Background Approach
2. Equilibrium Partitioning Approach (Water-Sediment and Biota-Water-Sediment Partitioning)
3. Apparent Effects Threshold Approach
4. Screening Level Concentration Approach
5. Spiked Bioassay Approach

The five approaches are discussed below and additional details can be found in the pertinent literature cited for each method.

Sediment Background Approach

In the Background Approach, sediment contaminant concentrations are compared to concentrations from reference background sites where contaminant levels are deemed to be acceptable (OMOE 1987, 1988). Using the Background Approach, levels are set according to a "suitable" reference site or "acceptable" level of contamination. A suitable reference site may be one where sediments are considered to be relatively unaffected by anthropogenic inputs. Alternatively a suitable reference site may be derived through sediment profiles. In the latter, the pre-industrial sediment horizon, as determined through techniques such as palaeontology, could be used to determine background levels. The basis of the Background Approach is the implicit assumption that concentrations above these background values have an adverse effect on aquatic organisms.

For the purposes of PSQG development a "pre-industrial" standard could be adopted only for metals. The strictly anthropogenic (man-made) organic contaminants, for which background levels should theoretically be zero, would require adoption of a contemporary surficial sediment standard, based on a suitable reference site.

Advantages

The data requirements of the Background Approach are minimal in that the method requires only the measurement of the chemical concentrations of contaminants in sediments. As such it can be used with the existing data, thus minimizing the need for additional data collection. The method does not require quantitative toxicological data and avoids the need to seek mechanistic chemical explanations for contaminant behaviour or biological effects.

Background limits have advantages from an enforcement perspective since the Background Approach does provide an indication of the chemical concentration for metals that is expected to occur naturally. While it is possible that biological effects may occur in some species at metal concentrations indistinguishable from non-anthropogenic background, it is difficult to justify enforcement of a standard that has never been realized in nature. Thus background levels for metals can provide a practical lower limit for management decisions. For organic contaminants, which are largely anthropogenic, background should theoretically be zero. In most areas, however, contaminants have found their way into sediment and a contemporary benchmark based on current average concentrations for a suitable reference area may provide the practical lower limit for enforcement.

Limitations

Since the Background Approach relies only on the chemical concentration of contaminants in sediments it has no biological basis. Because biological effects data are not considered, cause-effect relationships between sediment contaminant levels and sediment-dwelling organisms cannot be determined. The exclusive use of chemical data implies that sediment characteristics have no influence on the resultant biological effects, but rather that chemical concentrations alone are responsible for the observed effects. However, sediment characteristics (i.e., grain size, organic content, dissolved oxygen levels) have been shown to be major factors affecting benthic community composition (Gooday et al 1990; Death 1995; Remple et al 2000).

Implicit in the method is the assumption that the chemicals present are in their biologically available forms. The method therefore, makes no allowance for the occurrence of different chemical species with differing biological availability and toxicity.

A further limitation of this approach is that background levels tend to be highly site-specific. They therefore require the designation of a reference site, which itself is likely to be highly subjective.

Equilibrium Partitioning Approaches

Phase partitioning of organic compounds has been used to describe the distribution of certain organic compounds in aquatic compartments. Partitioning, like adsorption, is one of the processes by which organic compounds can be sorbed to sediments. A major difference however, is that partitioning is solubility dependent and therefore, reversible (i.e. equilibrium) partitioning of non-polar organic compounds is a function of their solubility in water. The very insoluble compounds, as a result, partition strongly to sediment with only very minor amounts in water. These compounds tend to have high partition coefficients, as measured by the octanol-water partition coefficient, K_{ow} . The K_{ow} is the ratio of the amount of the compound that is soluble in an organic solvent, such as octanol, relative to the amount soluble in water.

The partitioning approaches have been extensively investigated by the U.S. EPA (Pavlou & Weston 1984, Di Toro *et al*, 1991). A basic assumption of this approach is that the distribution of contaminants among different compartments in sediment is controlled in a predictable manner by a continuous equilibrium exchange among sediment solids and the interstitial water. Partitioning to these two phases can therefore be calculated by the quantity of sorbent in the sediment (for which organic carbon is the primary sorbent) and the partition coefficient K_{oc} . K_{oc} values, which can be estimated from K_{ow} , are normalized to sediment organic content.

The Equilibrium Partitioning approaches also assume that interstitial water is the primary route of organism exposure to contaminants in sediments. Therefore, this approach assumes that only the amount of contaminant partitioning to the water is of interest, the amounts partitioning to the sediments being considered as unavailable.

Using this approach, contaminant-specific partition coefficients are determined (generally expressed in terms of organic carbon content of sediment) and used to

predict the distribution of the contaminant between sediment and interstitial water. This approach, however, can only be used for contaminants that partition between environmental phases. Contaminants that do not partition appreciably into sediment organic matter, and those whose chemical behaviour is highly unpredictable (such as metals), cannot be considered using this approach.

Under the Equilibrium Partitioning approach, a generic (i.e. equally applicable to all sites) organic carbon-normalized partition coefficient K_{oc} is developed and is then multiplied by an existing water quality objectives/guidelines to derive a sediment guideline. In essence, the distribution coefficients for the non-polar organics are used to establish the chemical concentration in the sediments that, at equilibrium, will not exceed water quality objectives/guidelines in the interstitial water. Sediment Quality Guidelines based on the equilibrium partitioning of organics can be calculated in a number of ways, depending on the type of data available:

1. *Water-Sediment Equilibrium Partitioning Approach:*

The Water-Sediment Partitioning Approach is a generic partitioning method which derives a sediment quality guideline from the partitioning of a chemical to the water and the sediment solid phases. There is sufficient evidence to show that sediment organic carbon is the primary environmental factor influencing partitioning (Di Toro *et al.* 1985 in OMOE 1988). The partition coefficient, K_{sed} , is normalized for organic content and an organic carbon-normalized sediment-water partition coefficient is derived (K_{oc}). This can either be derived empirically, or calculated from the octanol-water partition coefficient. The partition coefficient is then multiplied by a water quality criterion (such as a water quality objective) to derive a sediment quality guideline.

2. *Biota-Water-Sediment Equilibrium Partitioning Approach*

The Biota-Water-Sediment Partitioning Approach is a generic partitioning method which derives a sediment guideline from an existing tissue residue criterion. It is a two step approach utilizing a generic water-biota bioconcentration factor (BCF) to relate the tissue criterion to a corresponding water concentration. For bioaccumulative substances this relationship determines the tissue-water concentration level (TWCL). The TWCL is the value that must not be exceeded in water in order to prevent exceedance of the tissue residue criteria from which the TWCL was derived. The TWCL, therefore, is equivalent to a water-quality criterion. Following this step the approach is similar to that described for the water-sediment approach with the TWCL used in place of the water quality criterion.

Advantages

Generic Partitioning Approaches are biologically based to the extent that existing water or tissue criteria are biologically based and, therefore, provide more defensible guidelines than the Background Approach. Since they make use of the virtual no-effect levels determined from existing Provincial Water Quality Objectives and Guidelines (PWQO/Gs) the sediment guidelines derived through generic partitioning approaches can be considered no-effect levels for the protection of those end-uses the water quality guidelines were designed to achieve.

The partitioning approach relies on an existing toxicological rationale which has been established during the development of the water quality criterion being used. Thus, a new toxicological evaluation is not required provided that the water quality criterion has been derived to protect those benthic organisms which are exposed to the interstitial water. However, a corresponding limitation to the approach is its applicability only to chemicals which have water quality criteria. Moreover, if the water and sediment criteria are meant to protect different organisms then an assumption is made that the two sets of organisms are of equal sensitivity to given levels of contaminants.

Limitations

The basic assumption that availability of an organic compound to aquatic organisms is controlled by the amounts partitioning to the water ignores both the sediments and food chain effects as potential sources. It has not yet been proven that the interstitial water is the only significant route of exposure and for the highly hydrophobic compounds (those with high K_{ow}); all of these sources may be significant routes of exposure.

Tissue residue criteria are generally based on human health considerations and human food consumption patterns (MacDonald 1994). Therefore, the tissue residue criteria apply to human food organisms such as fish, rather than benthic organisms. Similarly, the BCF applies to fish, and the water concentration (TWCL) thus derived applies to the water column in which the fish lives. This approach is limited by the substantial gap that exists between the water column compartment and the interstitial water compartment that is assumed to be in equilibrium with the sediments. The reduction in contaminant concentration from the interstitial water to the water column compartment is likely to be highly site-specific depending on local-circulation.

Current use of the Partitioning Approach is limited to those contaminants that exhibit predictable partitioning behaviour. Since the partitioning of metals in sediments is highly unpredictable (e.g., sediment-water partition coefficients for metals can span a wide range of values differing by orders of magnitude depending on such factors as redox potential, pH, dissolved oxygen and organic matter content of the sediment) and polar organics generally do not partition into sediment, the partitioning approaches are considered applicable only to non-polar organic compounds.

The scientific validity of a sediment guideline obtained through the partitioning approaches relies heavily on the accuracy of the partitioning coefficients (K_{oc}) used. The published values for partition coefficients obtained by different authors can differ by an order of magnitude. This presents great difficulty in choosing a representative value for use in guideline development work and unless a standard approach is used it will be difficult to obtain consistent or compatible guidelines using the EP approach.

At present the Equilibrium Partitioning Approach cannot account for all the forms a contaminant can exist in and all the possible sediment constituents it can partition to. This is currently a drawback to the Equilibrium Partitioning Approach since the various forms of a contaminant have their own toxicity and partitioning characteristics. Several species of a contaminant may be bioavailable and toxic, but often their concentrations are more or less linearly dependent on the concentration of a single species. While it has been possible to establish that one species correlates with the observed toxic effects for the non-polar organics, this has not been possible for the metals or the polar organics. The partitioning approach does not work for metals or polar organics due to

the multiplicity of adsorption mechanisms these undergo. It is not even clear which sediment components are controlling partitioning.

Apparent Effects Threshold (AET) Approach

The Apparent Effects Threshold (AET) Approach, as developed by Tetra Tech (1986) is a statistically based approach that attempts to establish quantitative relationships between individual sediment contaminants and observed biological effects. The biological effects can be both field measured effects such as changes in benthic community structure and laboratory measured effects through the use of sediment bioassays. The basis of this technique is to find the sediment concentration of a contaminant above which significant biological effects are always observed. These effects can be any or all of a number of different types, such as chronic or acute toxicity, changes in community composition, and bioaccumulation and are considered in conjunction with the measured sediment contaminant levels. Inherent in the approach is the assumption that observed effects above this level of contamination are specifically related to the contaminant of interest, while below this level any effects observed could be due to other contaminants.

Advantages

The AET approach is effects based and therefore more defensible than the partitioning approaches in relation to the protection of benthic organisms. The method assumes a direct cause-effect relationship between sediment concentrations of a contaminant and the occurrence of significant biological effects.

Unlike the partitioning approach the AET approach makes no assumptions regarding contaminant availability from the various environmental compartments. Therefore the effects on biota can be due to contaminants available through both adsorption from sediments and interstitial water and through absorption from ingested matter.

Limitations

The method is unable to separate the biological effects that may be due to a combination of contaminants. While assuming a cause-effect relationship, the method cannot clearly demonstrate a cause-effect relationship for any single contaminant. Thus, while definite ecotoxicological effects can be established, these cannot be attributed to any one chemical contaminant.

In using the AET approach care must be exercised in selecting the species of organism to be used and the particular type of effects (endpoints) to be considered. If the data used consist of mixed species and endpoints, the least sensitive of these will always predominate and the guidelines derived may not protect other more sensitive species. For example, if the data base for a particular contaminant contains data on acute toxicity to tubificid oligochaetes, then the AET will be designed to protect against acute toxicity to tubificids. It will not protect species that are more sensitive nor will it provide protection against chronic effects.

For most practical purposes this method requires chronic toxicity data since results from the existing database indicate guidelines tend to be higher than those calculated by

other means, in some cases by an order of magnitude. This is usually due to the use of acute toxicity data which needs a correction factor to adjust to chronic toxicity. The development of a chronic toxicity database (i.e., one based on reproductive effects and effects on the most sensitive life stages) itself requires a very extensive set of information which at present does not exist in a standardized form. In order to obtain such information, considerable laboratory testing will have to be carried out. In addition, for data from different investigators to be useful, consistency in procedures and definition of endpoints will be necessary.

In practice, guidelines generated by the AET approach are likely to be underprotective since this method determines the contaminant level above which biological effects are always expected. Biological effects, however, can be and are observed at chemical concentrations lower than these values, though these effects may not occur in all samples.

The AET approach is applicable for all types of contaminants, making use of both laboratory tests on sediments (spiked sediments) and field data. In laboratory tests of field-collected sediments it may not be possible to separate the effects of mixtures of chemicals. If spiked sediments are used, only single contaminant or known (specific) mixtures can be used and therefore this method suffers from some of the same limitations as the Spiked Bioassay method (discussed below). In using field collected sediments in conjunction with other field data (e.g. community composition), it is not possible to separate the effects of mixtures of contaminants and this method suffers from the limitations affecting the Screening Level Concentration Approach.

The Screening Level Concentration (SLC) Approach

The Screening Level Concentration (SLC) Approach, like the AET Approach, is an effects based approach applicable mainly to benthic organisms. The SLC approach uses field data on the co-occurrence in sediments of benthic infaunal species and different concentrations of contaminants. The SLC is an estimate of the highest concentration of a contaminant that can be tolerated by a specific proportion of benthic species. In its original derivation and application, the 95th percentile was used (Neff *et al* 1986).

The SLC, as developed by Neff *et al* (1986), is calculated through a two step process. First, for a large number of species (at least ten for each chemical) a species SLC (SSLC) is calculated by plotting the frequency distribution of the contaminant concentrations over all sites (at least ten) where the species is present. The 90th percentile of this distribution is then taken as the SSLC for that species (Figure 1a). The 90th percentile was chosen to provide a more conservative estimate of the SSLC. Extreme sediment concentrations may be an aspect of specific sediment characteristics resulting in low biological availability relative to the sediment concentration. By choosing the 90th percentile, these values are excluded. In the second step, the SSLCs for each species are plotted as a frequency distribution and the 5th percentile is interpolated from this distribution (Figure A1b). This is the SLC and represents the concentration which 95% of the species can tolerate.

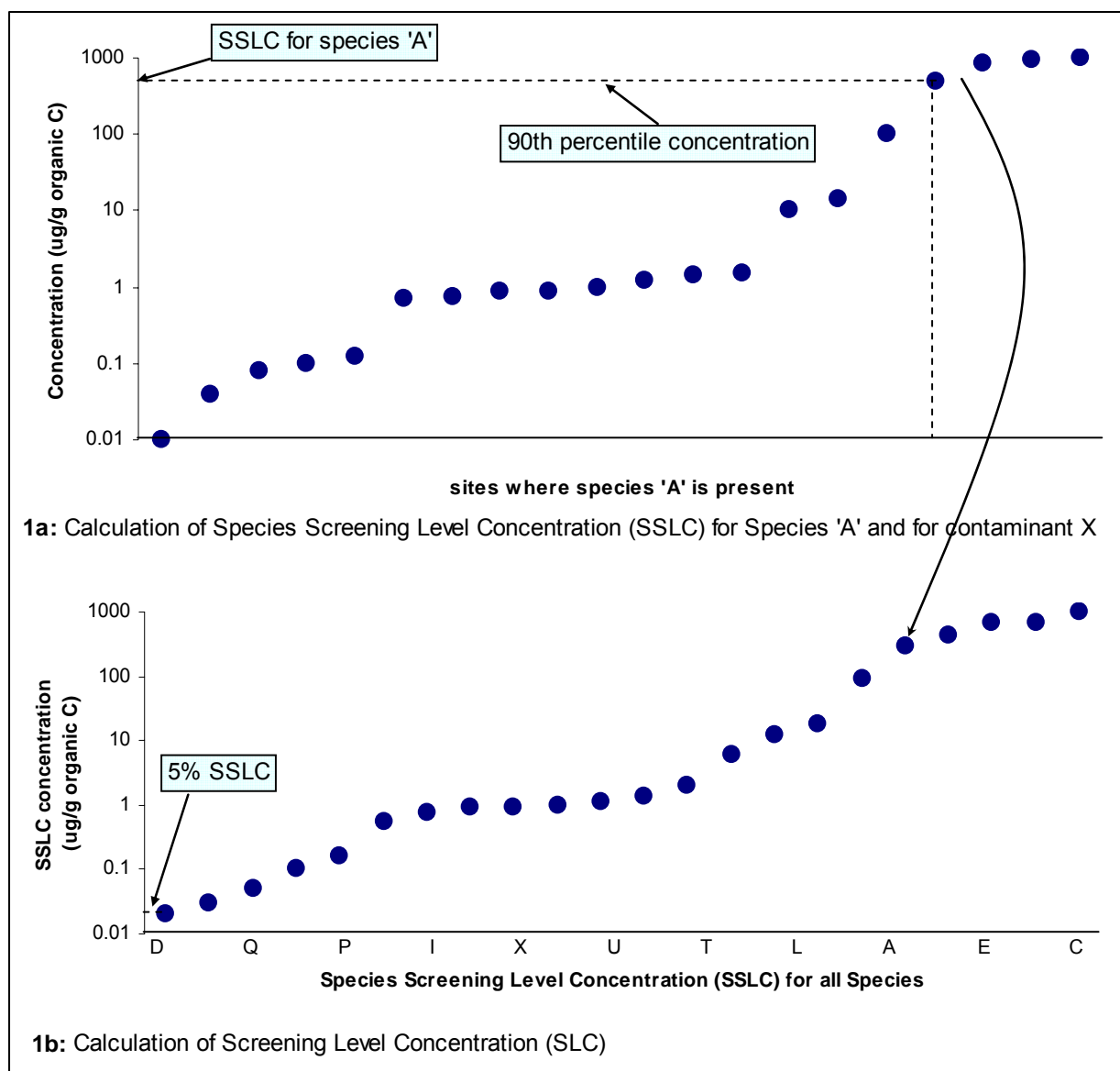


Figure A1. Screening Level Concentration Calculation

A basic assumption in the method is that the data cover the full tolerance range of each species. This assumption requires that a large range of chemical concentrations be sampled in each case (at least two orders of magnitude) since an SLC will be generated whether or not this assumption is true. This is important though sometimes difficult to verify. The difficulty lies in the fact that the full tolerance range of most species is not known.

Sediment contaminant concentrations for the non-polar organics are normalized to TOC content of the sediments. Since these compounds generally partition strongly to organic matter, the normalized concentration should more closely represent contaminant availability to benthic organisms. For metals and polar organics, bulk sediment

concentrations are used since the best normalization procedures for representation of metal availability are as yet unresolved.

Advantages

Since the SLC Approach does not make any assumptions about the absence of a species and considers only those species present, the SLC approach does not require a *priori* assumptions concerning cause-effect relationships between sediment contaminant concentrations and the presence or absence of benthic species. As no relationship is assumed it is not necessary to take into account the wide variety of environmental factors that affect benthic communities, such as substrate type, temperature and depth.

However, valid inferences can be drawn from this type of analysis regarding the range of sediment contaminant concentrations that can be tolerated by the sediment infauna since field data on the co-occurrence of benthic infaunal species and sediment contaminant concentrations are used.

Since the SLC Approach uses field data on the co-occurrence in the field of contaminants and benthic species, the environmental factors acting on the species distribution are already integrated into the data-set and the response determined is a measure of both the environmental factors and the contaminant levels. It also integrates changes in chronic responses such as reproduction/fecundity and sensitive life-stages, since it is a cumulative measure of effects. In addition, it integrates into the biological response any synergistic or additive effects from multiple contaminants as they would occur in natural sediments. Because of this, the SLC Approach overcomes the difficulties of applying bioassay data to field situations, and the lack of uncertainty associated with partition coefficients.

While it was originally developed primarily for use with non-polar organics (using TOC normalization) it is also appropriate for metals and polar organics as well since it can be used with or without TOC normalization.

At present the size of the database has determined that the SLC level be set at the 5th percentile of the SLC frequency distribution. However, as the database continues to expand it should be possible to reliably calculate the 1st percentile (i.e. the level of a contaminant that 99% of the species present can tolerate). The precision of the SLC is directly related to the size of the database and the range of variability of the various factors within the database. Therefore great care must be taken to include data taken over the full range of conditions since a database skewed to either lightly or heavily contaminated areas will yield guidelines that are either too conservative (overprotective) or do not provide adequate protection for aquatic life (under protective).

Limitations

The major limitation of the SLC Approach is the difficulty in determining a direct cause-effect relationship between any one contaminant and the benthic biota, since very rarely is a single contaminant present in natural situations. Therefore, the effects observed are related to the entire mixture of chemicals.

The range and distribution of contaminant concentrations and the particular species used to generate them can significantly affect the calculation of the SLC value. The use of only low values of contaminant concentration may not encompass the entire tolerance range of the species and the concentration would be below the level that would adversely affect the distribution of that species. In such situations, an SLC would still be generated but the value would be conservative and unrealistic. This can be overcome by ensuring that the database includes values from heavily contaminated areas.

The SLC is also sensitive to the species used in the database. Unlike the Partitioning Approach, the SLC Approach does not make any assumptions regarding the possible routes of effect from aquatic contaminants, all possible modes of exposure are taken into account. Since contaminant availability from the sediments may differ in relation to the feeding habits of the organisms used, the proportion of species from each of the feeding groups will determine the shape of the SLC curve. This can also be overcome by limiting the database to those organisms living in or feeding on the sediment.

Spiked Bioassay Approach

In this approach, dose-response relationships are determined by exposing test organisms, under controlled laboratory conditions, to sediments that have been spiked with known amounts of contaminants (OMOE 1987, 1988). Sediment quality guideline values can then be determined using the sediment bioassay data in a manner similar to that in which aqueous bioassays are used to establish water quality criteria. Where chronic toxicity data are not available, an approximation can be obtained by using acute toxicity endpoints that have been adjusted downwards by a factor of ten to obtain a chronic protection level and then applying a suitable safety factor.

Advantages

The major advantage of this approach is that a direct cause-effect relationship can be determined, at least under laboratory conditions, for a specific chemical or combination of chemicals for any species of organism.

Limitations

Despite this advantage, limitations exist that, at present, preclude the use of this method for setting guidelines. Techniques have not been standardized for spiking sediments and differences in methods/techniques can strongly influence the results. In addition, laboratory bioassays performed under controlled conditions may not be directly applicable to field situations where conditions may vary considerably from those encountered in the laboratory. In order to derive realistic guidelines from the Bioassay Approach efforts will have to be made to test different sediments with various chemical mixtures in differing proportions and using different organisms, as would exist in field situations.

Summary Evaluations of the Various Approaches to PSQG Development

The major objectives in the development of sediment quality guidelines are to provide protection to aquatic organisms and ensure water quality protection, as well as guidance in decision-making related to abatement efforts and remedial action. As such, they are intended to be both proactive and reactive in application. The primary basis for such decisions is the protection of biological resources against the lethal and sublethal effects of contaminated sediment.

The biological resources that could potentially be impacted by contaminants in sediment span a wide range. These include organisms that could be impacted directly, namely the benthic species that live in or feed on the sediment, and water column organisms that could sorb contaminants released from the sediment to water and/or through the consumption of benthic organisms; and those impacted indirectly such as non-aquatic consumers (humans and wildlife) of top aquatic predators such as fish.

In reviewing the five approaches to setting sediment guidelines, it is apparent that each approach has certain merits as well as limitations. The Background Approach while lacking a biological basis, does provide a good indication of the levels at which metals are expected to occur naturally and thus provides a realistic lower limit for guideline development.

The Partitioning Approaches to sediment guideline development use existing criteria such as a water quality or tissue residue criteria which can be considered as virtual no-effect values. The resulting sediment guidelines can therefore also be considered as virtual no-effect values for the protection of water column organisms from sediment-bound contaminants. The Partitioning Approach is attractive because it is capable of providing a measure of contaminant availability from sediments with a minimum of data. Due to the incorporation of various safety factors in the generation of PWQOs, this approach is able to provide an estimate of the no-effect level of a contaminant in sediments. How protective this value may be depends on the sediment organisms, the size of the safety factor, and the type of sediment. The approach is limited by its assumption of a single route of exposure for aquatic organisms and its restriction to the non-polar organics.

The AET Approach appears best suited to discriminating between contaminated and uncontaminated areas within a site, since the data used tend to be highly site specific. As a result, any guidelines derived will also be site-specific. The major limitation lies in the assumption of a cause-effect relationship that the methods prove unable to demonstrate. There is also a lack of chronic effects data suitable for AET applications, particularly if consistency in level of protection (i.e. single species and endpoint) is desired. Therefore, the AET Approach is judged less acceptable than the other effects-based approaches.

The SLC Approach has an advantage in that no cause-effect relationships are assumed and therefore, it does not need to account for all of the natural environmental factors that can affect organisms. The effects of these are already integrated into the data. The effects of multi-contaminant interactions are also factored into the data set used in the calculations and, with a sufficiently large database, the effects of other contaminants can be minimized. The SLC Approach would be less defensible on a theoretical basis than the Spiked Bioassay Approach if the data bases for the two approaches were

comparable. It has been found, however, that relevant information from bioassays is considerably lacking, especially in relation to the impacts of chemical mixtures on benthic populations. Due to the scarcity of Spiked Bioassay data, it is difficult to achieve consistency in the level of protection (i.e. a variety of species and endpoints must be considered). The problem could be rectified with further chronic data acquisition, particularly if standard spiking techniques were adopted. In practice, the methodology has not been standardized and variations in experimental protocol can greatly influence the results. The ability to transpose laboratory derived results to natural situations is also questionable.

Since there is presently a significant lack of adequate data for use in the development of sediment quality guidelines using the Spiked Bioassay Approach, the SLC Approach offers the best means of developing sediment quality guidelines for the protection of the benthic community. This is especially true since there a good database for the Great Lakes Region already exists.

In accordance with the merits and limitations of the various approaches to sediment guideline development, their use can be summarized as follows:

- Partitioning Approaches have been used to develop virtual no-effect levels for the protection of water quality and uses, and health risks associated with humans and wildlife through the consumption of fish. These can be used to set sediment contaminant levels that are also protective of these same uses.
- The Effects-Based Approaches (AET, SLC and Bioassay) are being used to develop guidelines for the protection of benthic organisms. Based on the existing information base, only the SLC approach is of immediate use in the development of sediment quality guidelines.
- The Background Approach has been used to establish levels where adequate data do not exist for application of any of the other methods or where the methods used are inappropriate for the type of compound. In addition, background levels provide a practical lower limit for management decisions.

As sediment bioassay techniques are refined and standardized it may be necessary to revise the protocol to accommodate these techniques as well, though it is unlikely that these will ever supplant field based approaches such as the SLC, since some field verification of laboratory results will always be necessary.

Data Requirements

A PWQO is required for setting levels according to the Partitioning Approach. In order to maintain consistency between sediment and water quality guidelines, levels set by other agencies will not be used.

At least three estimates of partitioning coefficients would be required to set a guideline using the partitioning approach. Guidelines based on fewer than the minimum number of estimates would be regarded as tentative.

The range of contaminant concentrations for the SLC calculations should span at least two orders of magnitude and include data from both heavily contaminated areas and relatively clean areas. Data from clean areas are needed to ensure that sensitive species are included in the SLC calculation, while heavily contaminated areas are needed to ensure that the full tolerance range of all the species is covered.

The database for the SLC calculations should be based on primarily benthic infaunal species and should minimize the reliance on epibenthic species. A minimum of 75% benthic infaunal species would be required to ensure that the observed effects are from sediment associated contaminants and not from water column effects.

Consistency in the species data used has to be ensured. This requires checking the data for synonymies, unusual species distributions, and level of identification. The minimum acceptable taxonomic level would be the genus, provided that species level identifications were also included in the data set from which the information was derived. Data using only generic level identifications could not be used.

The SLC database must include a large range of areas sampled in order to minimize the effects of unmeasured but co-varying contaminants. Since these are unlikely to occur in the same relation at all other areas, the effects of other contaminants can be reduced or excluded if a sufficiently large number of different areas are included.

A minimum of 10 observations are required to calculate an SSLC. A minimum of 20 SSLCs are required to calculate an SLC. This low number has been chosen so as not to exclude the less common or more sensitive species that may not be present at more highly contaminated sites.

Calculation of the Provincial Sediment Quality Guidelines

No Effect Level

Since this is intended as the level at which contaminants in sediments do not present a threat to water quality and uses, benthic biota, wildlife or human health, the parameter values used in deriving the No Effect Levels (NEL) must be the most stringent criteria.

The NEL is principally designed to protect against biomagnification through the food chain. Since these effects are most often observed with the nonpolar organics, this guideline level is not applicable to most of the trace metals. The partitioning approaches are used to set these guidelines since, with appropriate safety factors the PWQOs are designed to protect against biomagnification of contaminants through the food chain, as well as all water quality uses and organisms.

At present, reliable partition coefficients can only be derived for the nonpolar organics, since only these compounds undergo predictable partitioning behaviour in sediments. NEL Guidelines cannot be calculated for metals and polar organics.

Non-Polar Organics

The NEL for non-polar organics is obtained through a chemical equilibrium partitioning approach using PWQOs.

The calculations for each criterion are as follows:

The PWQO value is multiplied by an organic carbon-normalized sediment-water partition coefficient, K_{oc} . Normalization was recommended by Pavlou and Weston (1984) and OMOE (1988), since sediment organic carbon has been found to be the primary environmental factor influencing partitioning.

A PSQG is then derived through the equation:

$$SQG = K_{oc} \times PWQO/G$$

where PSQG is the sediment quality guideline normalized to the sediment organic carbon content (TOC). This is to a bulk sediment basis by assuming a 1% TOC concentration. A 1% level for sediment organic carbon is used for converting to a bulk sediment basis, since calculations using the SLC Approach have shown that this is the lowest effect level of organic carbon in the sediment. A bulk sediment calculation based on the actual organic carbon content of the sediment has been avoided for this reason.

The organic carbon-normalized partition coefficient is calculated from either an experimentally derived sediment-water partition coefficient:

$$K_{sed} = \frac{[X]_{sed} / o.c.}{[X]_{iw}}$$

where $[X]_{sed}$ is the concentration of compound X in the sediment (as mass of X/mass of organic carbon) and $[X]_{iw}$ is the concentration of the compound in the interstitial water (as gms/L) (Pavlou 1987) or it can be reasonably accurately derived from the octanol-water partition coefficient according to the formula developed by Di Toro *et al* (1985; in OMOE 1988).

$$\log_{10} K_{oc} = 0.00028 + 0.983 \log_{10} (K_{ow})$$

The K_{oc} value used is derived by taking the geometric mean of the available K_{ow} values.

Both measured and calculated K_{ow} values can be used to derive a K_{oc} and a number of values are required to estimate the K_{oc} used.

K_{oc} values should be calculated from laboratory derived sediment-water partition coefficients whenever possible, rather than from values derived from the octanol-water partition coefficient (K_{ow}).

Since the NEL criteria make use of the PWQOs, which employ safety factors to ensure conservative levels, it is anticipated that the sediment guidelines derived from these will be conservative as well. While the distribution of non-polar organics in the pre-colonial sediment horizon should technically be zero, it is recognized that a certain amount of

sediment contamination has occurred from remote sources through atmospheric inputs. Since guidelines set below these background levels would be impractical, the background levels must form the lower limits of any sediment quality guidelines. To this end, background levels for the non-polar organics are provided in this document for comparative purposes. These are based on the average of the upper Great Lakes, deep basin surficial (top 5 cm) sediment concentrations, or in some cases, on concentrations in bluff materials. It is expected that where the NEL criteria derived by the partitioning method fall below these background levels, the background levels will provide the practical lower limit for management purposes.

The deep basin surficial sediment concentrations from the upper Great Lakes can be considered as representative of atmospheric inputs of the persistent (generally nonpolar) organics. Table 4 gives the background levels for those compounds for which upper Great Lakes level have been calculated, and these can be considered as normal background levels for management purposes.

Lowest Effect Level

The Lowest Effect Level (LEL) is the level at which actual ecotoxicological effects become apparent. It is derived using field-based data on the co-occurrence of sediment concentrations and benthic species. The Screening Level Concentration Approach described in the previous section is used for all types of contaminants.

The calculation of the SLC is a two step process and is calculated separately for each parameter. In the first step, for each parameter the individual SLCs (termed Species SLCs) are calculated for each of the benthic species. The sediment concentrations at all locations at which that species was present are plotted in order of increasing concentration (Figure A1a). From this plot, the 90th percentile of this concentration distribution is determined. The 90th percentile was chosen to provide a conservative estimate of the tolerance range for that species. This would serve to eliminate extremes in concentrations that may be due to specific and unusual sediment characteristics. The 90th percentile is that locus below which 90% of the sediment concentrations fall.

In the second step, the 90th percentiles for all of the species present are plotted, also in order of increasing concentration (Figure A1b). From this plot, the 5th percentile and the 95th percentile are calculated. These represent the concentrations below which 5% and 95% of the concentrations fall.

Metals, Nutrients, and Polar Organics

Calculate the 5th percentile of the SLC based on bulk-chemistry sediment data. Since the guidelines are derived for province-wide application, the locations used should span a wide range of geographical areas within Ontario of varying sediment concentrations of the contaminant. It is important to ensure that both high sediment concentrations as well as low concentrations are used in the data set to ensure the result is not biased towards one end or the other, since this could bias the resulting SSLC. A minimum of 10 observations would be required to calculate a SSLC for any one species. This relatively low minimum has been chosen so as not to exclude less common species, or more importantly, the more sensitive species that may not be present at the more

contaminated sites and thus may not be represented at the majority of sites. A minimum of 20 SSLCs (i.e. 20 species) would be required for calculation of an SLC.

Non-polar Organics

Calculate the SLC as above, but using contaminant concentrations normalized to the organic carbon content of the sediments (i.e. mass of contaminant/mass of organic carbon as expressed by TOC).

The organic carbon normalized sediment contaminant concentrations are converted back to a bulk sediment concentration assuming a 1% TOC. A limit of 1% TOC has been imposed on the calculation since calculations using the SLC approach have shown that this is the lowest effect level of organic carbon in the sediment.

The Ministry also recognizes that certain parameters addressed in these guidelines, such as the trace metals, occur naturally in aquatic environments. In an area as geologically diverse as Ontario, natural sediment levels can vary considerably from one region of the province to another as a result of differences in local geology. Therefore, the Ministry realizes that certain sites will naturally exceed the LEL. In such cases, the local background levels, based on the pre-colonial sediment horizon, will form the practical lower limit for management decisions as described in the Implementation Section of this document.

Calculation of Site-Specific Background

Site-specific background is calculated as either:

- i) the mean of 5 surficial (top 5 cm) sediment samples taken from an area contiguous to the area under investigation, but unaffected by any current or historical point source inputs; or,
- ii) the mean of 5 samples taken by a sediment core from the pre-colonial sediment horizon. The pre-colonial horizon is generally determined as the sediment below the *Ambrosia* sediment horizon. Except in areas of high sedimentation, such as river mouths, this can be estimated as that sediment lying below the 10 cm sediment depth.

Severe Effect Level (SEL)

This level represents contaminant levels in sediments that could potentially eliminate most of the benthic organisms. It is obtained by calculating the 95th percentile of the SLC (the level below which 95% of all SSLCs fall).

Metals, Nutrients, and Polar Organics

Calculate the 95th percentile of all SSLCs using the bulk chemistry values.

Non-polar Organics

Calculate the SLC as for the metals, but normalizing the data to the organic carbon content (TOC) of the sediments. The TOC-normalized SLC is then converted to a bulk sediment value at the time of application to a specific site, based on the actual TOC concentration of the sediments at that site (to a maximum of 10%, the 95% SLC guideline for TOC).

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